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A comparison of logging and fire disturbance on biophysical attributes of the Northern jarrah forest

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A COMPARISON OF LOGGING AND FIRE DISTURBANCE ON BIOPHYSICAL ATTRIBUTES OF THE NORTHERN JARRAH FOREST

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Bachelor of Science (Hons)

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USE OF THESIS

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ABSTRACT

Ecologically sustainable forest management (ESFM) serves dual purposes: 1) to ensure the conservation of biodiversity, and 2) in production forest (forest managed for the production of timber and woodchips), to maintain an economically viable timber industry in perpetuity. A central axiom of ESFM is that any manipulation of a forest ecosystem should emulate the 'natural' disturbance patterns of the forest. This is based on the assumption that forest communities have evolved with endogenous disturbance regimes and will be better able to cope if exogenous disturbance (e.g., logging) remains within natural levels and spatial and temporal bounds of intensity. The objective of this thesis was to assess whether logging within one of Australia's temperate forest ecosystems, the jarrah forest, falls within the range of its natural disturbance regimes- and if not, whether logging reduces biodiversity within this ecosystem. As fire is the predominant natural disturbance in most Australian temperate forest, this objective was achieved by examining (i) whether there were initial differences between the impact of fire in unlogged forest compared with the impact of logging and fire in production forest and (ii) whether the biodiversity impacted by logging disturbance recovered within a period of time equivalent to that required for recovery from disturbance caused by fire.

Five management treatments were investigated at Mt Dale, located in the northern jarrah forest of Western Australia. Each treatment was replicated at four sites one hectare in size. Unlogged forest that had not been burnt for five years provided baseline data used for comparison with data collected in forest disturbed by fire and in forest disturbed by logging. The second treatment investigated fire disturbance on sites of unlogged forest that were burnt in 2000. The three remaining treatments investigated the initial and residual effects of logging in forest that was (i) gap logged and burnt in 2000 (contemporary logging practice involves logging and post logging burning), (ii) gap logged in 1990 and had not been burnt for five years, and, (iii) selectively logged prior to 1950 and had not been burnt for five years. These treatments allowed the objective of this thesis to be addressed by addressing three specific questions:

1) Are there differences in biophysical attributes in unlogged forest recently disturbed by fire compared with forest recently disturbed by gap logging and fire?
2) Do biophysical attributes recover 10 years after gap logging?
3) Do biophysical attributes recover 50 years after selective logging?

Sixty-seven attributes of forest structure were examined. Forest that was gap logged and burnt in 2000 had lower canopy cover, lower stand basal area (SBA) of trees and higher volumes of coarse woody debris (CWD) than unlogged forest that was burnt in 2000.

Similarly, forest gap logged in 1990 had lower SBA and higher volumes of CWD than unlogged forest. Structural changes caused by selective logging remained discernible after 50 years, with smaller trees and greater volumes of CWD recorded in selectively logged forest compared with unlogged forest. The slow growth rate of jarrah and slow decomposition of CWD suggest that structural changes will persist for centuries after contemporary logging disturbance in these forests.

The diversity of understorey plants, leaf litter spiders and birds was also investigated. When these taxa are combined to measure biodiversity, more species were recorded in unlogged forest that had not been burnt for five years than all other treatments. The assemblage of species in forest selectively logged prior to 1950 was more similar to the community occurring in forest gap logged in 1990 than unlogged forest. The two treatments burnt in 2000 had the lowest diversity of species, irrespective of logging. When the three taxa are considered separately, there were small differences in the diversity of plants and spiders in unlogged forest burnt in 2000 compared to forest gap logged and burnt in 2000. There were also less regenerating plant species and web-weaving spiders recorded in forest gap logged in 1990 compared to undisturbed forest. There were no statistical differences in the diversity of the three taxa between forest selectively logged prior to 1950 and undisturbed forest.

Contemporary silviculture by gap logging therefore does not emulate fire disturbance. Changes to biodiversity caused by this type of logging go beyond the temporal changes caused by a single fire event. This study shows that gap logging needs to be recognised as a disturbance that causes short-term, and potentially long-term, changes to jarrah forest biodiversity. In contrast, forest selectively logged prior to 1950 appears to have similar biophysical attributes as unlogged forest. Given that only 9% of the jarrah forest is currently unlogged, stands that contain these attributes could be managed to ensure the comprehensive, adequate and representative conservation of biodiversity across the jarrah forest.

Declaration

I certify that this thesis does not, to the best of my knowledge and belief:

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CHAPTER 1

Thesis Overview

1.1 Thesis Overview

The conservation, promotion and appropriate management of biodiversity are thought to be imperative to future human and non-human welfare. Biodiversity is defined to include genes, species, populations, communities, and landscapes, with each level of biological organisation exhibiting characteristic composition, structure and function (Noss 1990; Heywood 1995). Biodiversity provides humankind with services such as clean air and water, healthy soil and plants, cures for disease, and protection against increased temperatures that result from the 'enhanced greenhouse effect' (Ehrlich & Mooney 1983; Pimental *et al.* 1997). These services result from millions of years of evolution. Technological innovation has not, as yet, provided alternatives. Along with the moral, spiritual, cultural and economic reasons for conserving biodiversity, humankind now recognises that their own subsistence is directly related to conserving the world's biodiversity.

Forests are a major repository of biodiversity. These ecosystems provide habitat for approximately 65% of the world's terrestrial taxa (World Commission on Forests and Sustainable Development 1999). Among other services, forests regulate local and global climate, and influence the Earths' energy budget (Ullsten *et al.* 1998). Since forests are major carbon reservoirs in the global carbon cycle (Dixon 1994; Carey *et al.* 2001; Pregitzer & Euskirchen 2004), their preservation helps to mitigate carbon dioxide emissions and the resulting temperature increases (Pinard & Cropper 2000). Forests also retain soil and protect watersheds from salination, siltation and increased levels of nutrients (Pearce 2001). Moreover, thousands of species likely to have useful medicinal properties are found in forests (Balick & Mendelsohn 1992). The importance of

conserving forest biodiversity has been highlighted in many forums including the 'Earth Summit' (United Nations Conference on Environment and Development 1992), the 'Santiago Declaration' (Anon 1995) and the 'Millennium Report' (Reid *et al.* 2005).

Despite these values, humans have eliminated a significant proportion of the world's forests and the biodiversity found within them. At least 40% of all forests are estimated to have been removed in the last millenia (World Commission on Forests and Sustainable Development 1999). Species extinction rates are estimated to be between 100 - 1000 times that which naturally occurs, with an untold number of species likely to have already become extinct as a result of deforestation (Myers 1979; Whitmore & Sayer 1992). Moreover, deforestation has significantly impaired ecological services, including carbon cycling (Shutou & Nakane 2004), regional hydrology and climate (Laurance *et al.* 2000), and also threatens populations of medicinally important plants (Dawson & Powell 1999). Despite increased awareness of the importance of conserving forest biodiversity, a growing human population, the possibility of climate change, and the increased need for forest-based resources have lead to projections of large shifts in biodiversity in remaining forests (Laurance *et al.* 2000; Hansen *et al.* 2001). It is estimated globally that 16 million hectares are deforested per annum, representing approximately 0.4 % of the world's forest cover (Matthews 2001 in Pearce 2001). Based on this rate of deforestation, most of the world's intact forest ecosystems will be degraded and fragmented within the next century, and much of the current biodiversity found in forest ecosystems will be lost. Ecologists have an obligation to identify and restrain these predicted losses (Ehrlich & Ehrlich 1981; Kati *et al.* 2004).

This thesis assesses one human activity that can result in a loss of forest biodiversity. The removal of trees for timber and pulp (a practice hereafter referred to as "logging") has affected forest biodiversity in most populated regions on Earth. While the consequence of logging tropical forests has received much media attention, the impact of this disturbance on temperate forest biodiversity should be of equal concern (As 1993; Bader *et al.* 1995; Lindblad 1998). In comparison with tropical forests, temperate ecosystems have had a larger part of their area lost due to clearance, and their removal, modification and degradation continues at an unprecedented rate in many regions (Norton & Kirkpatrick 1995). Although there may be general agreement with the notion temperate forests can be logged while biodiversity is sustained, there is concern that this has not been achieved in many regions (e.g., Recher *et al.* 1980; Spurr *et al.* 1992; Hansen *et al.* 1993; Haila *et al.* 1994; Imbeau *et al.* 2001; Watson 2004). There is little evidence as to whether periodic logging in forest ecosystems significantly affects biodiversity, and even less is known as to whether any changes could be mitigated through appropriate logging methods (Summerville & Crist 2002).

Australian temperate forests have been logged since European settlement (1788). Since this time, more than half of these forests have been removed for timber, agriculture and urban settlement (RAC 1992). A large proportion of the remaining forest estate is now managed for timber and pulp production. Many ecologists believe that current logging practices in these forests could adversely affect biodiversity (e.g., McIlroy 1978; Recher et al. 1980; Kirkpatrick & Bowman 1982; Davey & Norton 1990; Clarke 1992; Lunney 1992; Calver et al. 1995; Lindenmayer 1995; Norton 1996; Lindenmayer & Recher 1998). Others

argue that current logging practices do not threaten biodiversity because sufficient areas are reserved from logging, and appropriate measures taken within production forests to mitigate the effects of logging (e.g., Abbott & Christensen 1994, 1996). Governments depend on the quality of advice they receive from ecologists since they face the pressure to guarantee, in principle, the long-term stability of forests. They know they must balance the needs of forest-based industries with conservation of biodiversity.

The Regional Forest Assessment and Agreement Programme is the most recent attempt by governments to resolve the contentious issues relating to forest management and represents the largest planning activity related to the environment undertaken in Australia (Dargavel 1998). This programme began with the signing of the National Forest Policy Statement (NFPS) between the Commonwealth and all State governments except South Australia in 1992 (Commonwealth of Australia 1992). Ten Regional Forest Agreements (RFAs) were established to cover all Australian forests containing large timber industries. Each RFA set down a framework for management for 20 years within a designated region, and attempted to resolve the 'forest debate' by:

a) providing for the long-term stability of forests and forest based industries;
b) creating a Comprehensive, Adequate and Representative (CAR) reserve system; and
c) implementing ecologically sustainable forest management (ESFM).

This thesis focuses on ESFM. A central axiom of ESFM is that any manipulation of a forest ecosystem should emulate the 'natural' disturbance patterns of the region that existed prior to 'industrial' manipulation, such as logging (Burrows *et al.* 2002a). A 'disturbance' is defined as any relatively

discrete event in time that disrupts ecosystem, community, or population structure, and changes resources, substrate availability, or the physical environment (White & Picket 1985). Ecologists recognise that periodic disturbances are essential to the maintenance of forest biodiversity (Pickett & White 1985; Hunter 1999; Haeussler *et al.* 2002). Disturbances destroy, transform, transport and create attributes of forest structure (Hansen 1999), driving the cyclic renewal of community composition and releasing resources and promoting nutrient cycling (Attiwill 1997).

ESFM is based on the assumption that forest biodiversity has evolved with endogenous disturbance regimes (disturbances that are part of the natural system, and part of autogenic development), and will be better able to cope if exogenous disturbance (disturbances that are introduced to the system, and part of allogenic development) remains within natural bounds of spatial and temporal severity (Franklin 1989; Gillis 1990; Hansen *et al.* 1991; Haila *et al.* 1994; Attiwill 1997; Stork *et al.* 1997; Peterken 1999; Mitchell *et al.* 2002). Exogenous disturbances such as logging may result in the loss of biodiversity if this action exists beyond the boundaries determined by previous natural disturbance regimes (Norton 1996; Attiwill 1997; Lindenmayer & Recher 1998). 'Boundaries' include the amount of area that is disturbed, the intensity of the disturbance event, and the frequency between disturbance events (Miller 1982; Stork *et al.* 1997; Huston 1999; Summerville & Crist 2002). In most Australian forests it is not known whether logging exists within the range of natural disturbance. The aim of this thesis is to assess whether the severity of logging lies within the range caused by natural disturbances that occur within one of Australia's temperate forest

ecosystems (the jarrah forest), and, if not, whether logging affects biodiversity within this ecosystem.

Natural disturbances include storms, fires, droughts and tree-falls. In jarrah forest, the most regular and widespread natural disturbance is fire (Hopper 2003). Fire is a keystone ecological process (*sensu* Holling 1992) because it has helped determine the structure and composition of species for millions of years. Natural fire regimes have been caused by lightning strike since the late Tertiary (Singh *et al.* 1981). In regions adjacent to the jarrah forest, such fires are predicted to have occurred at intervials of 30 to more than 100 years (Hassell & Dodson 2003). This fire regime changed significantly with the human habitation, which began at least 50 ka BP (Hassell & Dodson 2003). The burning practices of Indigenous people are considered 'natural' agents of disturbance given the length of time they have lived in the jarrah forest (Taylor 1990; Watson 2005 in Appendix 1). Although very little is known of the actual burning practices of Indigenous people in this period, it is believed fires were used to create, conserve, and exploit fine-grained habitat mosaics (Hallam 2003). As a result of Indigenous burning practices, fires are thought to have increased in frequency, and decreased in intensity (Jones 1969; Bowman 2003). Although it is doubtful that lifeforms in the jarrah forest have adapted to fire *per se* (Main 1996; Main 2001; Hopper 2003), different properties of fire (e.g., patchiness, frequency, intensity and season) are important factors that contribute to the mosaic of life forms now found in the jarrah forest (Van Huerck *et al.* 1999; Burrows & Wardell-Johnson 2003).

Logging has been a significant disturbance since 1829, when Europeans settled the southwest of Western Australia. It is estimated that 2 783 950 ha (~90%) of

jarrah forest has been logged in the last 150 years (Conservation Commission of Western Australia 2004). Concern about the sustainability of this industry in the jarrah forest has existed for over a century (Royal Commissions of 1877 and 1903 in Calver & Wardell-Johnson 2004). A large timber industry continues to exist within this ecosystem, logging ~ 15 000 hectares every year (Department of Conservation and Land Management 2003, 2004a). Significantly, it is not known whether logging lies within the bounds of natural disturbance in jarrah forest, or whether there will be long term losses of biodiversity as a result of logging disturbance. Three specific questions will be addressed in this thesis:

1) Are there differences in biophysical attributes in forest recently disturbed by fire compared with forest recently disturbed by gap logging and fire?
2) Do biophysical attributes recover 10 years after gap logging?
3) Do biophysical attributes recover 50 years after selective logging?

This thesis has a conventional structure (Fig.1.1). The chapter (Chapter 2) introduces the reader to the biophysical characteristics of jarrah forest, the history of logging in this ecosystem, and the specific aims and scope of my research. Chapter 3 outlines the design of this study, the location of study sites, and the selection of indicator taxa. Chapter 4 describes how forest structure is influenced by fire and logging disturbance. Chapters 5-7 assess how logging and fire affects the species richness and assemblage of understorey plants (Chapter 5), ground spiders (Chapter 6) and birds (Chapter 7). Chapter 8 assesses whether there is a discernible relationship in diversity among the three taxa. The thesis concludes with a discussion of the significance of this research in relation to current management practices (Chapter 9).

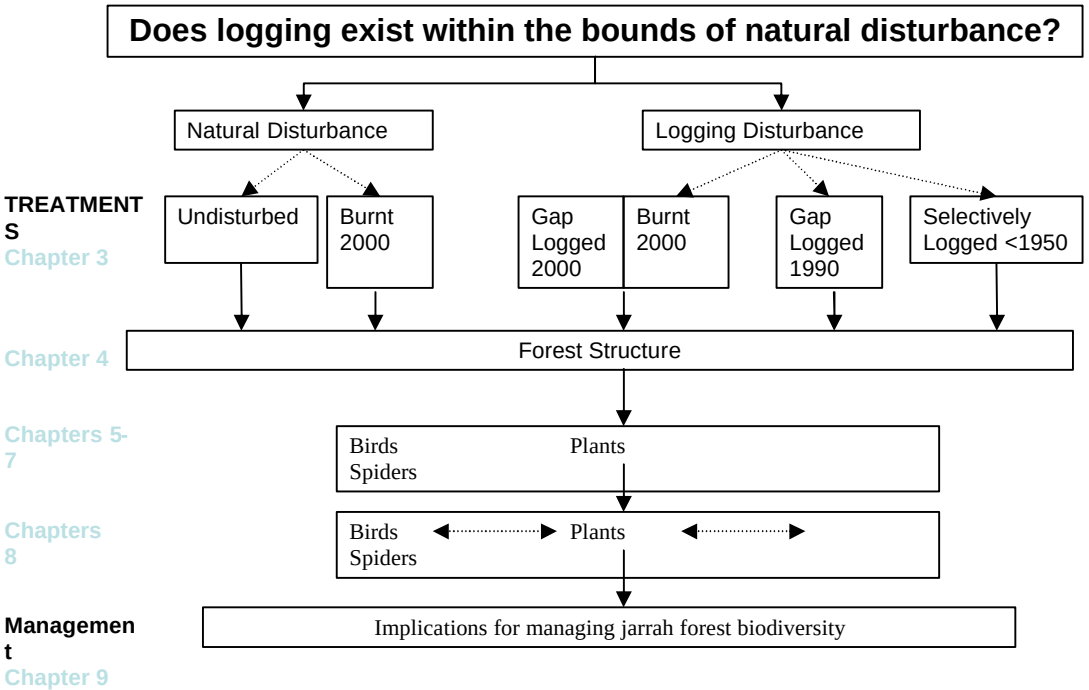


Fig. 1.1 A conceptual model of the design of this thesis.

CHAPTER 2

The Jarrah Forest

2.1 The jarrah forest

The suite of ecosystems that make up the jarrah forest are found only in the southwest corner of Western Australia. Their distribution is a subset of the much larger distribution of the Western Australian endemic *Eucalyptus marginata* Sm. (jarrah) (Wardell-Johnson *et al.* 1997) (Fig. 2.1). Fringing the coast, jarrah grows as a short-boled tree within *E. gomphocephala* DC. (tuart) dominated woodland (*sensu* Specht *et al.* 1974). In the east of its range, jarrah grows as a multiple-stemmed mallee within woodland communities dominated by *E. wandoo* Blakely (wandoo) and *E. accedens* W.Fitzg. (powderbark wandoo) (Beard 1990). At the centre of its east-west distribution, jarrah grows as a large tree that dominates forest in the north (the jarrah forest), and to the south occurs in *E. diversicolor* (Benth.) L.A.S.Johnson (karri) dominated forest. In the jarrah forest, *Corymbia calophylla* K.D.Hill & L.A.S.Johnson (marri) co-occurs with jarrah, but rarely reaches the same density (Bell & Heddle 1989). Jarrah dominated forests represent approximately 80% of the 3.38 million hectares of temperate forest that existed in Western Australia prior to European settlement (1829) (Conservation Commission of Western Australia 2004).

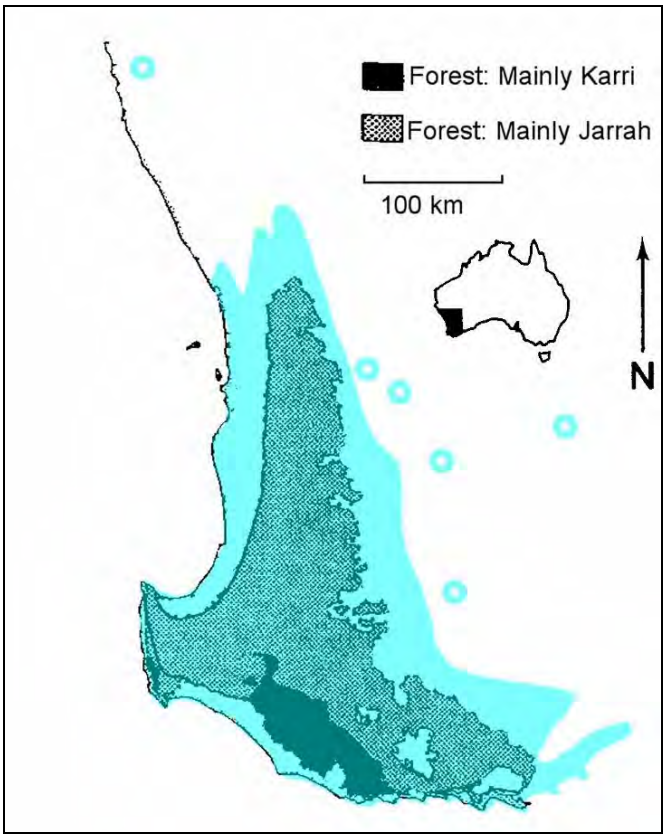
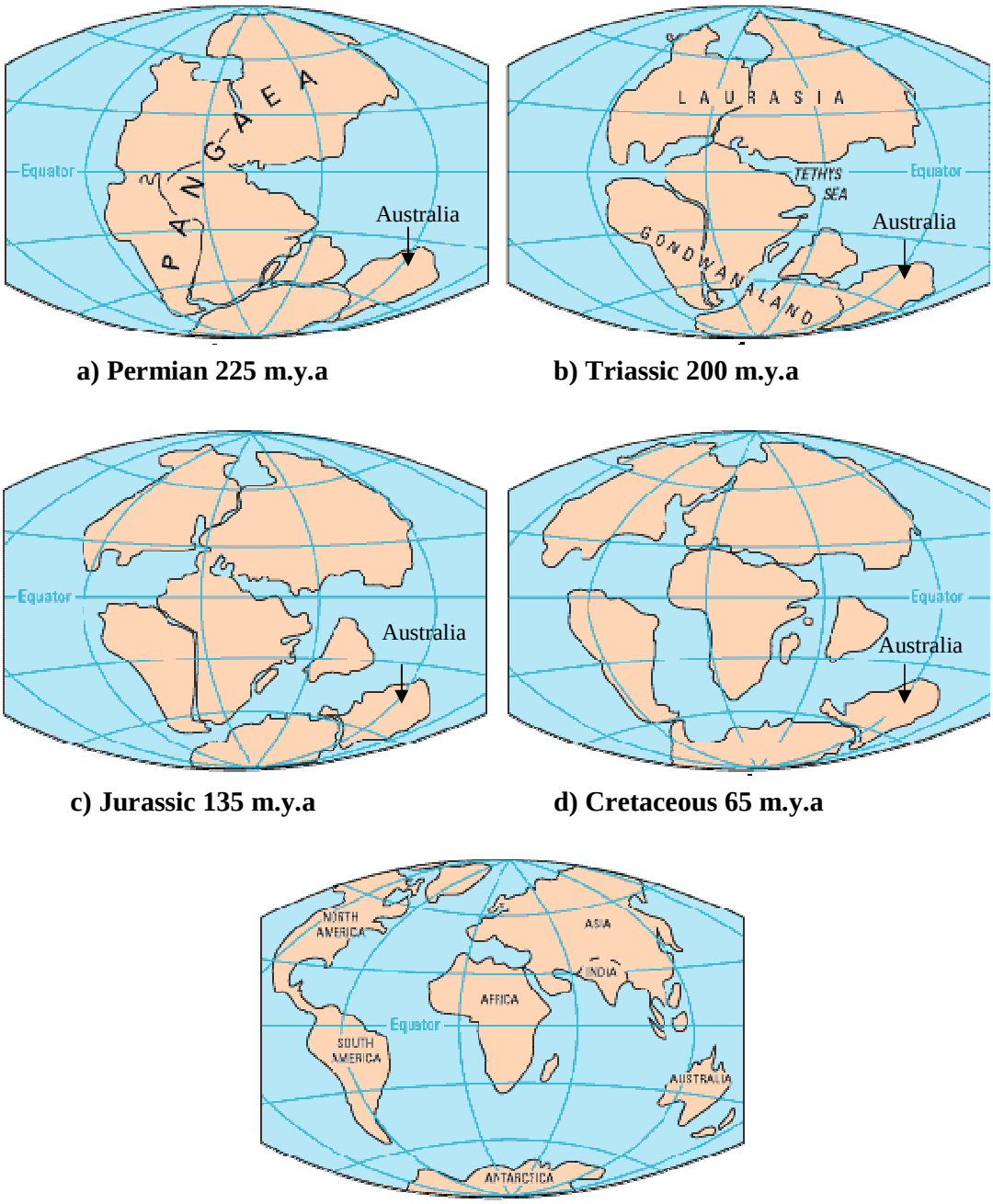


Fig. 2.1 The distribution of the jarrah plant (shaded) (modified from Dell & Havel 1989 p.6) overlaying the distribution of temperate forest in the southwest of Western Australia (modified from Wardell-Johnson & Nichols 1991 p. 162). Shaded circles represent outlying populations of the jarrah plant. Temperate forests are broadly divided into two forest types, defined on the basis of the dominant eucalypts ***Eucalyptus marginata*** ('jarrah') and ***E. diversicolor*** ('karri') (Burrows & Wardell-Johnson 2003).

It needs to be immediately recognised that several ecosystems that are not jarrah forest exist within the shaded area demarcated as 'mainly jarrah' (Fig. 2.1). These ecosystems include heath-like vegetation that occurs on granite outcrops, and woodlands dominated by marri and wandoo that occur on the scarp and the slopes of deeply excavated, small valleys (Beard 1990). Riparian forests comprising river gums and paperbarks also exist along some watercourses (Abbott & Loneragan 1986). The vast majority of the area, however, is dominated by the jarrah plant and hence considered 'jarrah forest'.

The biodiversity of the jarrah forest differs from other temperate forests due to the unique evolutionary processes to which the southwest of Australia has been exposed (Wardell-Johnson & Horwitz 1996, 2000). These processes are the result of plate tectonics, the movement and isolation of landmasses, and changes in climate. Approximately two hundred millions years ago (m.y.a), all the world's continents formed one super continent, Pangaea, of which the Australian sector was located near the South Pole (Hopper *et al.* 1996) (Fig. 2.2). Soon after, Pangaea split to form Gondwanaland (made up of modern day Australia, India, Africa, South America, and Madagascar) and Laurasia (Bradshaw & Weaver 1992). By 60 m.y.a, Gondwanaland had also divided, and South America, Africa, India and South America had all drifted north towards to the equator (Heatwole 1987). In contrast, Australia was still connected to Antarctica, and remained unglaciated, above sea level for almost 200 million years and had no significant mountain building events (Hopper *et al.* 1996). This situation provided unparalleled opportunities for unique terrestrial taxa to evolve (Norton 1996).



a) Permian 225 m.y.a **b) Triassic 200 m.y.a**

c) Jurassic 135 m.y.a **d) Cretaceous 65 m.y.a**

e) Present Day

Fig. 2.2 a-e The changing positions of continents over the last 225 million years (modified from Bradshaw & Weaver 1992). Early Australia is identified with an arrow.

In the Tertiary period (5 - 66 mya), Australia broke away from Antarctica and drifted north (Attiwill 1997). The Australian landscape changed from lush, humid rainforest, dominated by cycads, podocarps and conifers; to rainforest dominated by taxa that thrived in drier climates (e.g., Myrtaceae) (Archer 1996; Hopper & Gioia 2004). The onset of aridity in central Australia in the late Tertiary (approximately 5 mya) imposed climatic boundaries to the north and the east of forest in the southwest of Western Australia. The evolution of the 'Nullabour Plain' and deserts north of this region then isolated the southwestern biota from southeastern congeners (Hopper *et al.* 1996).

Most of the species that occurred in the rainforests of late Tertiary became extinct in the Pleistocene (Archer 1996). As the climate became drier and less humid, rainforests were replaced by eucalypt-dominated sclerophyll forests (Hopper *et al.* 1996). These forests were considered 'wet forests' because they occurred in the 'humid' (Semeniuk 1995), or 'high rainfall' zone (Hopper 1992) of southwestern Western Australia, as well as in similar climates in eastern Australia. However, periods of aridity were so severe in the southwest of Australia that wet forests were driven southwestwards, which first isolated them from the north, and then forced them off the edge of the continent, or into remnants too small to retain most of the 'old endemics' of wet forests (Archer 1996). Where many rainforest lineages survived in eastern Australian temperate forest, only those capable of surviving in small areas with moist refugia and microhabitats persisted in Western Australia (Hopper *et al.* 1996).

The last time climate-induced contraction in forest distribution in southwest Australia is thought to have occurred 4 – 6 000 years ago (Semeniuk 1995).

Since then, the high rainfall zone in southwest Australia has expanded (Fig. 2.3). Forests spread throughout this zone, and are now the only forests on Earth that grows in Mediterranean[1] climates without human intervention (Dell & Havel 1989). Compared to forest areas of similar size in eastern Australia, the jarrah forest has a much lower number of mammal species (Archer 1996) and approximately half the bird species (Abbott 1999). This low diversity can be explained in part by this relatively recent expansion of the temperate forest in southwest Australia (Archer 1996). Most of the mammal and bird species in the jarrah forest are thought to have invaded from drier central Australian regions (Archer 1996). Compared to eastern Australian temperate forests, the biota of the southwest forests are also impoverished in other taxa including mosses (Stoneburner *et al.* 1993).

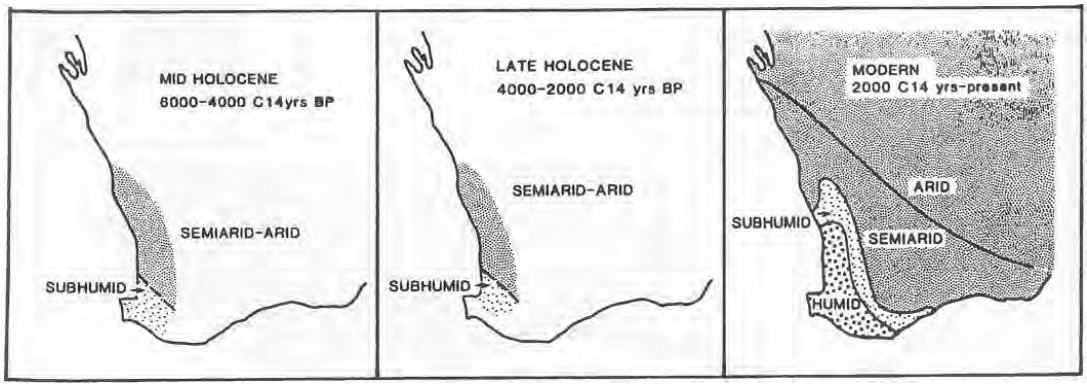


Fig. 2.3 Reconstruction of climate for the middle to late Holocene in 2000-year intervals (modified from Semeniuk 1995). Note the Holocene subdivisions (*viz.*, mid Holocene, late Holocene and Modern) are informal, and refer to intervals used in Semeniuk's study.

[1] Mediterranean climates are those that have pronounced wet and dry seasons, with precipitation mostly at the cooler time of year. Outside Australia, similar climates occur in southern Africa, southern California, central Chile, and the entire region around the Mediterranean Basin. Sclerophyll shrublands normally occur in these climates (Dell & Havel 1989).

Despite the relative low diversity of some taxa, the jarrah forest is within a region (the Southwest Botanical Province) considered one of the world's 34 biodiversity hotspots (Brennan *et al.* 2004) containing many endemic species of

vascular flora and fauna of low vagility (Abbott 1995; Hopper ***et al.*** 1996; Wardell-Johnson & Horwitz 2000). For example, this region contains at least 8000 species of vascular plant, of which 75% are endemic to southwest Australia (Hopper 1992). This represents almost half of the plant species in Australia (Hopper 1992). The high diversity is thought to be due to factors including i) the antiquity of some landforms and floral lineages, ii) the complex soil mosaics in the subdued landscape, iii) dynamic climate change during the Quaternary, and iv) the evolution of fragmented genetic systems, which led to heterozygous progeny despite the inbreeding caused by environmentally induced small populations (Hopper ***et al.*** 1996; Main 2001; Hopper & Gioia 2004).

Both karri and jarrah forests contain high levels of biodiversity and endemicity (Hopper ***et al.*** 1996; McKenzie ***et al.*** 1996; Judd 2004). However, there are broad differences in their structural and biological diversity due to differences in rainfall, nutrient availability, and disturbance regimes (Wallace 1957; Commonwealth of Australia and Western Australian State Government 1998). Karri forests are considered 'wet sclerophyll' forests because they are limited to areas of relatively high rainfall (> 1100 mm per year) (Commonwealth of Australia and Western Australian State Government 1998). The distribution of karri is also closely related to presence of younger soils and topography (Bradshaw & Lush 1981). In contrast, the jarrah forest is a 'dry sclerophyll forest' that occurs on the extensively laterised landscape of the Darling Plateau. This plateau is renowned for its impoverished soils and relatively low rainfall (Dell & Havel 1989; Churchward & Dimmock 1989; Hingston ***et al.*** 1989). The jarrah and karri forest are recognised as distinct temperate forests, and managed

using different management practices (Commonwealth of Australia and Government of Western Australia. 1999).

The term 'jarrah forest' reflects an incorrect perception that the forest is homogeneous (Wardell-Johnson & Horwitz 1996), when in fact, stands vary in size and structure throughout the 'jarrah forest' region. On the basis of canopy density and height, jarrah forest is classified as open forest in the north, and closed forest in the south (Specht ***et al.*** 1974). The height of dominant trees also declines significantly along a strong west to east gradient of decreasing rainfall (Abbott & Loneragan 1983a; Davison 1997). Towards the western edge of its distribution, high quality forest is present towards the western edge of the jarrah forest distribution, with the height of co-dominant trees exceeding 25m (Davison 1997). Inland towards the east of its range, tree height ranges from 15 - 24m (Abbott & Loneragan 1983; Abbott ***et al.*** 1989). The understorey contains even greater diversity, with 21 different plant communities estimated to exist in the northern parts of the jarrah forest (Havel 1975a; 1975b), and 17 in the southern parts (Strelein 1988). These communities represent a complex multidimensional continuum of species, responding to a number of environment variables including topography, available moisture, soil nutrients, and specific fire regimes (McCutcheon 1980; Havel 1989; Groves & Hobbs 1992; Burrows & Wardell-Johnson 2003).

This structural and biological diversity led to several attempts to delineate areas of jarrah forest based on its structure and species composition. These sectors include the 'prime' and 'non-prime' jarrah forest as defined by Lane Poole (1921) (in Abbott & Loneragan 1986), the 'high' and 'low' productive forests as

defined by Abbot & Loneragan (1983a) and the 'Northern Jarrah Forest' and 'Southern Jarrah Forest' recognised by Beard (1990). Beard's (1990) delineations were recognised in the Interim Biogeographic Regionalisation for Australia ('IBRA') (Department of Conservation & Land 2004b). The Western Australia Regional Forest Agreement used a more detailed classification system, recognising twelve different ecosystems (Fig. 2.4).

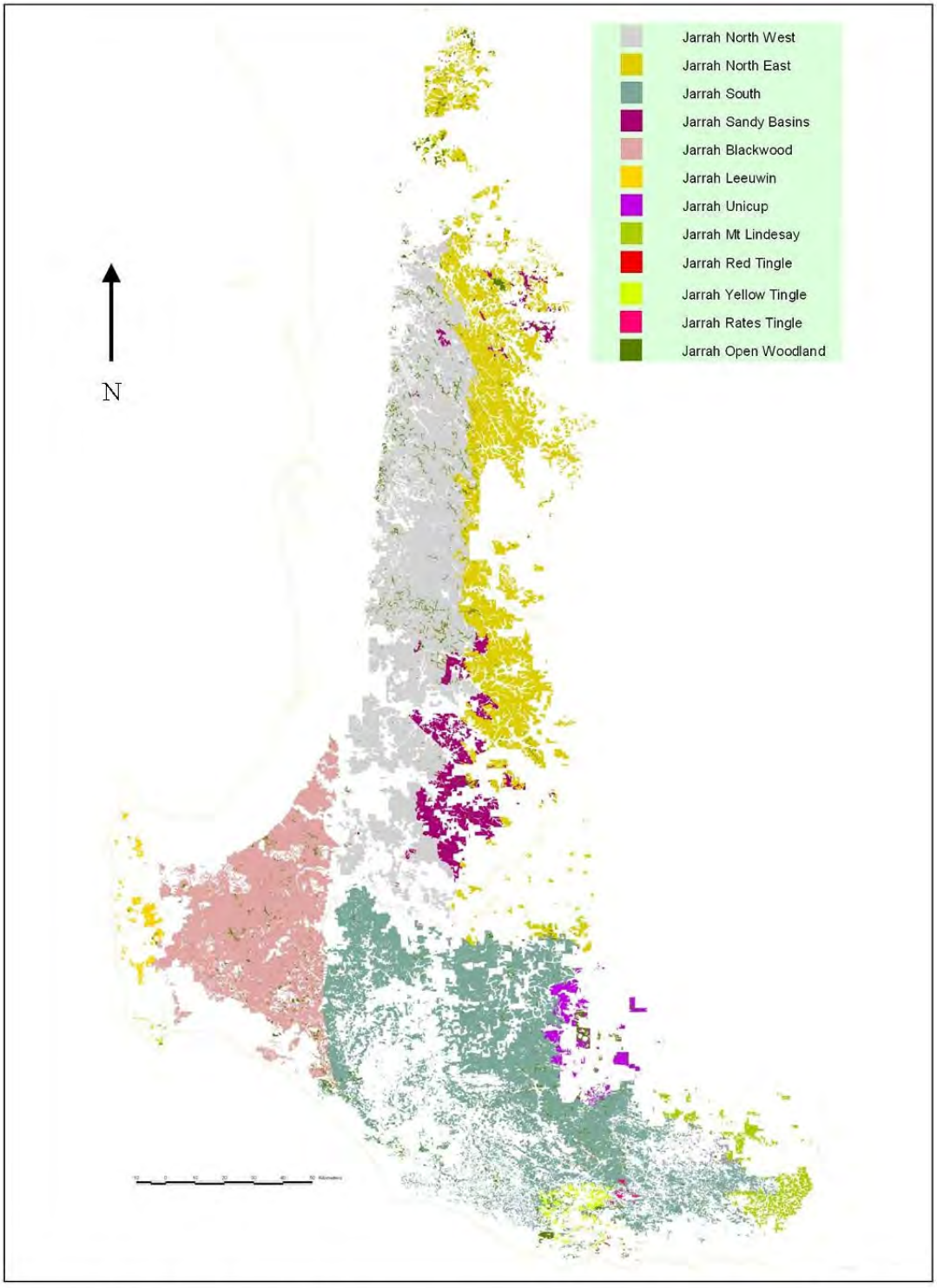


Fig. 2.4. The twelve jarrah forest ecosystems recognised in the Western Australian Regional Forest Agreement (1999) (Modified from Commonwealth of Australia and Western Australian State Government 1998).

2.2 Human activity in the jarrah forest

Very little is known of the activities of Indigenous people in the jarrah forest prior to European settlement in Western Australia (1829). Aboriginal culture did not include written language, and although drawings and symbols were used to supplement speech, there are no written histories to serve as data repositories prior to European arrival (Pearce 1989). Despite this, the frequent, low intensity burning practices of Indigenous people are thought to have altered the composition and structure of jarrah forest vegetation (Hallam 1975; Bowman 1998; Abbott & Burrows 2003; Hassell & Dodson 2003). The hunter-gatherer lifestyle and low density of Aboriginals across the forest landscape meant that other human disturbances, such as hunting, had minimal impact on jarrah forest biodiversity (Pearce 1989).

European settlement affected jarrah forest biodiversity by introducing several new disturbances (Hobbs 1996). An estimated 35 % of the jarrah forest has been cleared for urban development and agriculture (Commonwealth of Australia 1998), and many exotic animals, plants and micro-organisms have been introduced, both deliberately and accidentally. Some of these species must now be controlled by vast government and private expenditure (Hopper ***et al.*** 1996). One such example is the European fox, ***Vulpes vulpes***, which has caused the local extinction of several mammals (Burbidge & McKenzie 1989; Abbott & Christensen 1996; Christensen 1997a). To conserve remaining mammal populations, a program named 'The Western Shield' was launched in 1996 to control foxes by exploiting the natural tolerance of native fauna to sodium monofluoroacetate, commonly known as the poison 1080 (Department of Conservation and Land Management 2004b). Native fauna can tolerate low-doses of 1080 because of co-evolution with plants of the ***Gastrolobium*** genus,

which naturally contain varying amounts of flouroacetate, the toxic principle of 1080 (King 1990). In areas where foxes have been controlled, native mammals such as rock-wallabies, possums, woylies and numbats have increased in abundance (Department of Conservation and Land Management 2004b).

A phenomenon known as jarrah 'dieback' is another new disturbance that has caused the loss of biodiversity in jarrah forest. This name conjures up the symptom of declining tree crowns, but it is also used to describe groups of jarrah deaths, scattered jarrah deaths, and patch death of a large number of mid- and understorey species (Davison 1997). The first detailed description of dying patches came from field work carried out in 1947 and 1948 (Hamilton 1951). Since then, the name 'jarrah dieback' has been used for several disorders which may be interrelated (Davison 1997). The most frequently cited cause jarrah dieback is the introduce soil borne oomycete *Phytophthora cinnamomi* (Podger 1972; Dell & Malajczuk 1989). The disease kills hundreds of different species of plants, including jarrah, and severely alters the structure of the forest (Postle 1986; Dell & Malajczuk 1989). The families Proteaceae and Epacridaceae are particularly susceptible, and a number of species are already on the verge of extinction because of the disease (Hopper *et al.* 1996; Shearer *et al.* 2004). Jarrah deaths may also be caused by waterlogging, or by a combination of waterlogging damage and *P. cinnamomi* infection (Davison 1997). The phenoma known as 'dieback' has spread throughout the southwest of Western Australia (estimated to exist within 15% of the jarrah forest), and strategic actions have been implemented to limit further spread (Dell & Malajczuk 1989; Department of Conservation and Land Management 2004b). These actions include prohibiting

vehicles from travelling on unsealed roads after rain, and washing all vehicles before and after entering infected sites.

A third new disturbance in the jarrah forest is mining. Alcoa of Australia Limited has carried out bauxite mining and rehabilitation in the jarrah forest since 1963 (Roche *et al.* 1997). Approximately 450 ha of forest are cleared, mined and rehabilitated each year (Grant & Lonergan 1999a). The ore occurs in 'pods', which usually range from between 2 and 60 ha in size (Nichols & Nichols 2003). Disturbance by mining causes large changes to the structure of the forest, and impacts on the richness and composition of fauna and flora communities (e.g., Wykes 1985; Bellairs & Bell 1993; Koch & Ward 1994; Brennan 2002). The rehabilitation of mined areas has become a focus of study in jarrah forest because rehabilitation after bauxite mining aims to re-establish a self-sustaining forest (e.g., Roche *et al.* 1997; Ward *et al.* 1997; Grant & Loneragan 1999a,b; Ward 2000; Grant 2003; Nichols & Nichols 2003). These studies have lead to several changes in the techniques used to re-establish mine sites which include the return of pre-existing topsoil to mine sites and the treatment of all broadcast seeds with aerosol smoke to enhance germination rates (Ward *et al.* 1997; Roche *et al.* 1997) The current rehabilitation practices of Alcoa are successfully re-establishing over 80% of the plant species present in the pre-mining jarrah forest (Grant & Loneragan 1999b).

The fourth new disturbance in the jarrah forest is logging. Since European settlement (1829), it is estimated that at least 90% of the jarrah forest has been disturbed by logging (Commonwealth of Australia and Government of Western Australia 1998). The limited amount of research into the impact of logging in

jarrah forest means that this disturbance could be having a significant impact on biodiversity without being recognised (Calver & Dell 1998a,b; Calver & Wardell-Johnson 2004). The unique structural and biological diversity of jarrah forest means research conducted in other temperate forest may not be directly transferable to this region. Therefore, specific research is required in jarrah forest to design appropriate management strategies that adequately conserve biodiversity (Burrows *et al.* 1994; Morris *et al.* 1996; Stoneman *et al.* 1997; Calver & Dell 1998b).

2.3 Logging the jarrah forest

Jarrah is an aesthetically pleasing, dense, hard and durable as a result of a very tight grain structure (Wallace 1957). When Europeans first settled in the Swan Region (1829), it was quickly identified as a useful timber and a logging industry was established almost immediately (Mills 1989). However, records of logging prior to 1918 are incomplete due to insufficient government supervision at this time (Abbott & Loneragan 1986; Herbele 1997). The number and location of stumps show foresters selectively logged over large areas. High quality forest around local trading routes was virtually clear-felled, whereas less productive forests were logged relatively lightly (Herbele 1997). During this time, half of the areas in the north and central jarrah forest were logged twice, with some areas logged up to five times (Wardell-Johnson & Calver 2004). It is estimated that over 400 000 ha of forest were logged during this period (Wallace 1965 in Calver 2003).

Growing public pressure against waste led to Royal Commissions in 1877 and 1903, and the establishment of the Forests Department in 1918 (Wardell-Johnson & Calver 2004). Logging practices since 1918 have been reviewed many times with reliable timber harvesting records in the form of maps and reports being kept by the Forest Department (now held by the Western Australian Department of Conservation and Land Management) (e.g., Abbott & Loneragan 1986; Havel 1989b; Herbele 1997; Stoneman ***et al.*** 2005). These reviews show that for the first 50 years of its existence, the primary objective of the Forest Department was to produce timber and protect water supplies (Abbott ***et al.*** 2003). Using a variety of silvicultural practices, most of the unlogged forest that remained was logged during this period (Havel 1989b; Stoneman ***et al.*** 1989b). In addition, some of the previously logged forests were thinned to promote the growth of the most profitable trees (Abbott ***et al.*** 2003).

Explicit consideration of biodiversity conservation, forest reserves, mining, recreation and water catchment protection was first reflected in a Forestry Department Working Plan in 1972 (Calver & Wardell-Johnson 2004). Prior to this, the selection of the reserve sites was more often a matter of personal fancy than an objective process (Rundle 1996). Although the amount of forest set aside for conservation increased in the 1970s (Havel 1989a), when the Department of Conservation and Land Management (DCLM, formerly CALM) replaced the Forest Department in 1984, almost all publicly owned jarrah forest (State Forest) consisted of stands of various ages regenerating from logging of varying intensity. The Conservation and Land Management Act (1984) stipulated that all forests in Western Australia were to be managed for multiple purposes, including conservation, recreation, water and timber production. The CALM Act (1984)

brought new logging practices that had been previously trialled to ensure the adequate re-growth of the jarrah forest. These practices are complex and contained in Silvicultural Silvicultural Specifications 3/90, 1/91, 1/95, and 4/97 (Department of Conservation and Land Management 1990, 1991, 1995 & 1997). A simplified version is presented below.

The jarrah forest that can be logged (the production forests) is divided into forest blocks. When a forest block is selected to be logged, it is divided into a series of coupes. Coupes are the management unit for forestry operations, where one coupe is logged at a time. Prior to logging, a lignotuber survey is conducted to determine the amount of potential regeneration present. Within a 'patch' (defined as a discrete area, ranging from 1 to 200+ ha) one of three silvicultural treatments are then employed depending on the regeneration potential*:*

1. '**Thinning**'- aims to increase the growth of selected crop trees. Crop trees are those with a well-developed crown, or the potential to form a well-developed crown, and a bole capacity capable of producing high quality timber. Other trees are removed to reduce competition, thereby allowing crop trees to grow vigorously into higher value products. Wherever possible, patches are thinned in preference to 'regeneration release' or 'establish regeneration' techniques which are discussed below.

2. **'Regeneration release' -** is sought where there are insufficient crop trees to merit thinning, and where the stocking of ground coppice and saplings is likely to lead to regeneration after harvesting. Regeneration release is also known as "release regeneration", "gap creation" and "gap logging" (hereafter referred to as gap logging). The objective of gap logging is to remove the

overstorey to allow the regeneration of understorey coppice into crop trees. To prevent extensive alteration to large areas of forest, gap logged patches are a maximum of 10 ha in size, and separated by Temporary Exclusion Zones (TEAs, also known as 'coupe buffers') that are 100-150m wide. Gap logging is the most intensive silvicultural treatment (Abbott *et al.* 2003).

3. **'Establish regeneration'** – occurs where there is an inadequate stock of crop trees for thinning, and insufficient ground coppice and saplings available for gap logging. Establish regeneration (or shelterwood) involves the partial removal of the canopy and competing understorey rootstock species, burning to create an ashbed, and spreading seed or planting where poor natural seed stocks exist. Seedlings are encouraged to establish and develop into ground coppice by reducing the competition of overstorey and understorey species. Some canopy is maintained until ground coppice is developed and can respond to gap logging techniques.

Although not recognised in Silviculture Guideline 1/95, a fourth silvicultural practice (Single Tree Selection) is practised in some stands. Single tree selection is conducted by retaining vigorous trees and creating small gaps by removing less vigorous trees (Department of Conservation and Land Management 1997). Burrows *et al.* (2002a) suggested that the current definition of single tree selection is inadequate.

It is estimated that approximately 15 000 hectares are logged annually by any of these four practices (Department of Conservation and Land Management 2003, 2004a). This represents about 1.5 % of the available forest (Craig 1999). For all

silvicultural practices, the jarrah forests are not logged on any rotation length *per se.* However, calculations of sustained yield are based on variable rotations ranging from 100-220 years, with the majority of forest being logged every 170 years (Ferguson *et al.* 2003). Between these rotations, the jarrah forest can be thinned to remove competing vegetation. These rotations are designed so that approximately 40% of the jarrah forest is dominated by mature and senescent stages of development, 40% by immature trees, 15% by juvenile trees and 5% by establishment stages of development (Commonwealth of Australia and Government of Western Australia 1998). These classes were not defined in the Comprehensive Regional Assessment (1998) (Commonwealth of Australia and Government of Western Australia 1998).

2.4 Fire as a silvicultural tool

European arrival quickly resulted in the end of Indigenous burning practices. Bringing with them European concepts of naturalness, the new managers of the forest quickly attempted to arrest the 'unnatural' disturbance caused by fire. The conservator of forests in 1916, Lane Poole, believed the total exclusion of fire would enable natural succession to proceed, resulting in less undergrowth and a less flammable forest (pers. comm. Wardell-Johnson 2003). Until 1955, all efforts were made to exclude large fires in the forests. Although wildfires still occurred in some areas, the exclusion of regular fire is thought to have resulted in a build-up of large volumes of woody debris (Stewart 1951 in McCaw & Burrows 1989). In the early 1950s it became apparent that the fire exclusion policy was not practicable as the accumulation of fuels in protected compartments produced fires which were inevitably beyond the capacity of fire crews to control (Bell *et al.* 1989). The policy of fire exclusion was finally

abandoned after several highly destructive fires in the 1960s (McCaw & Hanstrum 2003). It was replaced with prescribed burning (Christensen & Abbott 1989), in which efforts are made to burn 70% of the most productive forest every 5-7 years (Wardell-Johnson & Nichols 1991).

Prescribed burning is now undertaken across the forest for many purposes, including fuel hazard reduction (for protection of forest and community assets), site preparation (for forest regeneration, and for maintaining or enhancing biodiversity values), regulating undesirable vegetation (Burrows 1985; Bradshaw *et al.* 1991) and maintaining biodiversity in the long-term (Burrows & Wardell-Johnson 2003). These prescribed fires are usually undertaken in autumn or spring, with the resulting fires of low to moderate in intensity (Christensen & Abbott 1989). Occasionally, very destructive wildfires occur during the summer months which cause much larger amounts of damage (e.g., kill trees), but these stand replacing events are rare (Burrows *et al.* 2002a). In areas disturbed by mining, fire is used to increase species similarity between rehabilitated areas and native forest by stimulating the germination and resprouting of fire adapted species (Grant & Loneragan 1999a,b). With respect to silviculture, logged areas are burned before and after logging (Calver & Wardell-Johnson 2004). Post logging fire is designed to promote regeneration of jarrah and remove excess woody debris (Department of Conservation and Land Management 1997; Hobbs 2003). Contemporary silviculture in jarrah forest therefore consists of two disturbances: (i) logging - the removal of timber, and the creation of space to allow for the next generation to develop, and, (ii) fire – post logging to promote the regeneration of jarrah.

2.5 Conserving jarrah forest biodiversity

2.5.1 The development of a reserve system

Nature reserves are a cornerstone of strategies for conserving biodiversity (Hansen & Rotella 2002). The primary role of reserves is to provide habitat that enables the maintenance of natural evolutionary processes, and to sustain populations of species that would otherwise be jeopardised by logging (Lindenmayer & Franklin 2003). The signing of the Western Australian Regional Forest Agreement (1999) saw the implementation of a Comprehensive, Adequate and Representative ('CAR') reserve system in accordance with the JANIS[2] (1997) criteria. Fifteen percent of forest that existed prior to European arrival was reserved in 'CAR Formal', 'CAR Informal' Reserves and other areas of Public land protected by prescription. CAR Formal Reserves represent dedicated reserves as described by JANIS criteria and include National Parks, Conservation Parks and other reserves (e.g., Section 5(g) CALM Act Reserves) that are secure, and require action by the Western Australian Parliament for revocation. CAR Informal Reserves are areas in production forest permanently protected from logging (Bradshaw 2002; Conservation Commission of Western Australia 2004). Informal reserves include River and Stream Zones (60-400m wide corridors buffering streams), Travel Route Zones (100- 200m corridors that extend each side of major travel routes), and Diverse Ecotype Zones (rock outcrops, wetlands, heath, sedge, herb and low density woodlands). Formal and informal reserves are managed under 'natural' disturbance regimes, that is, they are burnt every 5-7 years and trees are allowed to die naturally.

[2]The nationally agreed criteria for the establishment of a CAR reserve system for Australian forest identified the following objectives for biodiversity conservation (JANIS 1997):

- to maintain ecological processes and the dynamics of forest ecosystems in their landscape context,
- to maintain viable examples of forest ecosystems throughout their natural ranges,
- to maintain viable populations of native forest species throughout their natural ranges, and
- to maintain the genetic diversity of native species.

Since signing the Western Australian RFA, the reserve system has been modified to include all remaining 'old growth' forests (Government of Western Australia 2001). Old growth forest are defined as (from JANIS 1997 p. 14):

> Forest that is ecologically mature and has been subjected to negligible unnatural disturbance such as logging, roading and clearing. The definition focuses on forest in which the upper stratum or overstorey is in the late mature to over mature growth phases.

These forests are conserved primarily in formal conservation reserves (National Parks, Nature Reserves or Conservation Parks). Patches of old growth (hereafter referred to as 'unlogged') forest that are too small to be included in the formal reserve system are now conserved in informal reserves within production forest (~ 6% of all old growth forest).

The current reserve system is not large enough to ensure the ongoing persistence of all biodiversity found throughout the jarrah forest (Department of Conservation and Land Management 2004b). When new legislation passes through Parliament, an estimated 14% of the northern jarrah forest and 18% of the southern jarrah forest will be protected in reserves (Conservation and Land Management 2004). Successful conservation of forest biodiversity requires the appropriate management of non-reserved forest (the 'matrix') and the establishment of a system of reserves (Norton & Kirkpatrick 1995; Hobbs 1996; Lindenmayer and Franklin 1997a,b; Mitchell & Craig 2000). In jarrah forest, over 50 % of remaining forests can be logged (Commonwealth of Australia and Government of Western Australia. 1999). Managing the production forests is therefore an essential supplement to maintenance of conservation reserves because production forests can: (i) determine the size, viability and spatial dispersion of populations, (ii) facilitate the movement of organisms, (iii) buffer

sensitive areas such as reserves from the impacts of severe disturbance, and, (iv) maintain the integrity of aquatic systems (Lindenmayer & Franklin 2003).

2.5.2 Conserving biodiversity in production forests

When my research was undertaken, jarrah forests were managed under the 1994 – 2003 Forest Management Plan (Department of Conservation and Land Management 1994). Under this plan, conservation in production forests consisted of implementing an informal reserve system (primarily designed to protect riparian habitat and rare vegetation complexes, as well as to limit visibility of logging operations within the forestry block) and protecting some attributes within coupes that are logged. These attributes included primary and potential habitat trees, and logs. Primary habitat trees were defined as those trees that were mature or senescent and are > 70cm d.o.b. (d.o.b. is a standard measure of the diameter over bark of a tree at 1.3m above the ground), and that have indications of being useful for fauna. In all silviculture treatments, four primary habitat trees per hectare were protected. Furthermore, in gap logged patches, 6 to 8 'potential habitat trees' (trees that are immature to mature, 30-70cm d.o.b., and have the potential to develop habitat) per hectare were protected from logging. One ground habitat log (between 30 - 100cm in diameter and at least 1.5m long), or stump, per hectare was retained even if it showed no obvious sign of use.

The signing of the Western Australian Regional Forest Agreement (1999) initiated large changes to forest management practices in relation to conserving biodiversity in jarrah forest (Commonwealth of Australia and Government of Western Australia. 1999). Under this agreement, all forest (reserved and unreserved) is to be managed in line with principles of ecologically sustainable

forest management (ESFM). In 2000, the Conservation and Land Management Act was amended and the following five principles of ESFM were implemented (Sections 19):

a) That the decision-making process should effectively integrate both long-term and short-term economic, environmental, social and equitable considerations (the 'sustainability' principle),

b) That if there are threats of serious or irreversible environmental damage, the lack of full scientific certainty should not be used as a reason for postponing measures to prevent environmental degradation (the 'precautionary' principle),

c) That the present generation should ensure that the health, diversity and productivity of the environment is maintained or enhanced for the benefit for future generations (the 'intergenerational' principle),

d) That the conservation of biological diversity and ecological integrity should be a fundamental consideration in decision making (the 'biodiversity' principle),

e) That improved valuation, pricing and incentive mechanisms should be promoted (the 'efficiency' principle).

The sustainable management of jarrah forests has been a subject of controversy among scientists that work on these ecosystems. Managers of production forests argue that this resource is being sustainably managed (e.g., Abbott & Christensen 1994; 1996), and that some principals of ESFM (such as the 'precautionary principle') have been followed since the formation of the Forests Department in the 1920s (Lee & Abbott 2003). Other scientists conclude that logging has not been sustainable (e.g., Calver & Wardell-Johnson 2004). Calver & Wardell-Johnson (2004) compared increments of growth (m^3) with the amount of timber (m^3) being removed through logging to show that past logging has been unsustainable. There is concern that insufficient data hide the possibility that logging is having a substantial impact on biodiversity (Calver & Dell 1998a,b).

2.6 Understanding logging impacts on jarrah forest biodiversity

A comprehensive investigation of the short-term impacts of logging practices in jarrah forest (the 'Kingston project') was initiated in 1993 by the Department of Conservation and Land Management (described in detail by Burrows *et al.* 1994 and Abbott *et al.* 2003). This project examined the immediate impact of two silvicultural prescriptions (Gap and Shelterwood logging) on plants, mammals, ground arthropods and birds using a BACI (Before, After, Control, Impact) experimental approach (see Abbott *et al.* 2002; Burrows *et al.* 2002b; Strehlow *et al.* 2002; Abbott *et al.* 2003). Their research showed that although most plants and animal populations were not found to be significantly affected by logging in the short term, some communities were affected by logging for at least six years (more detailed reviews of these findings are given in the introduction of chapter 5-7). The Kingston project is a long-term project, and provides an excellent basis to understand the ongoing impacts of logging. However, given the extent to which the jarrah forest has been logged, and the fact that it continues to be disturbed by new forms of this disturbance, retrospective research was needed to assess whether logging has had any significant long-term impacts on biodiversity.

2.7 Aims

As outlined in Chapter 1, the objective of this study was to assess whether disturbance by logging in the jarrah forest lies within the bounds of the impacts caused by fire; and if not, whether this affects biodiversity. The research specifically addressed this objective by assessing whether logging exists within the temporal bounds of natural disturbance: that is, whether biodiversity recovers in an equivalent amount of time after logging disturbance as it does after fire

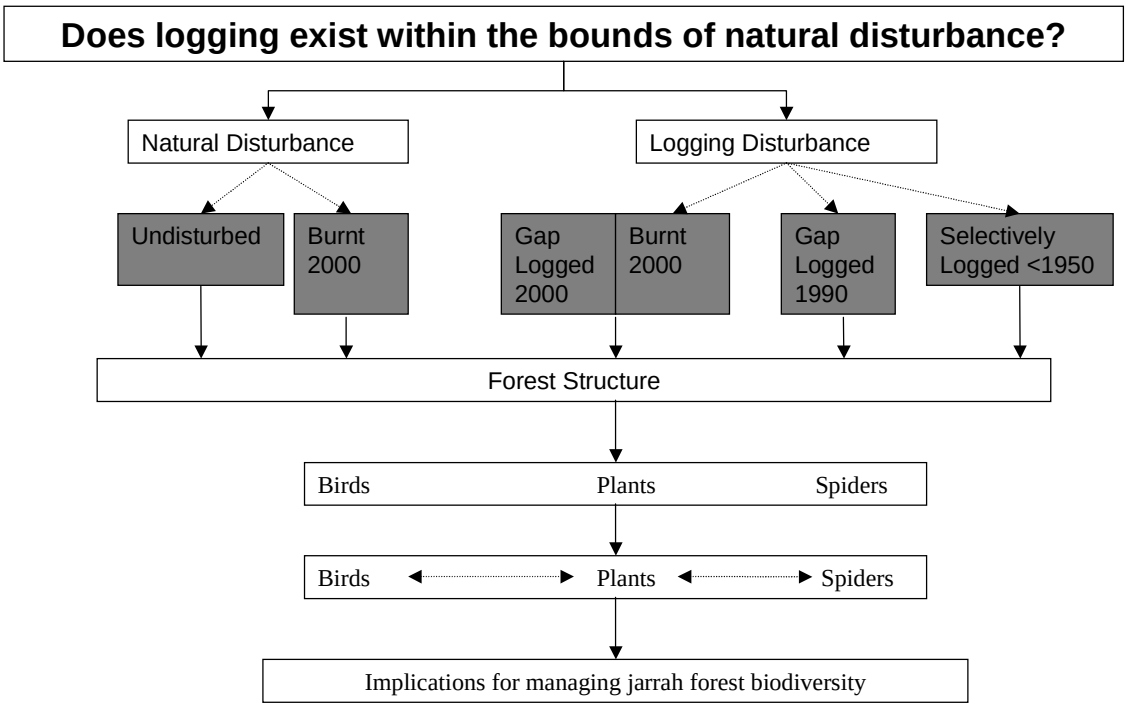
disturbance. The ability of a community to return to its pre-disturbance composition is referred to as resilience (Westman 1978). As the Department of Conservation and Land Management attempts to burn the entire forest estate every 5 -7 years, it is assumed that biodiversity in jarrah forest is resilient and recovers within 5-7 years after fire.

This objective was achieved by assessing two aims. The first examined whether there were initial differences between the impact of fire in unlogged forest and the impact of logging and fire in production forests. The second aim examined whether the biodiversity affected by logging disturbance recovered in a similar period of time as that caused by fire disturbance. These two aim were addressed by assessing three questions:

1) Are there differences in biophysical attributes in forest recently disturbed by fire compared with forest recently disturbed by gap logging and fire?
2) Do biophysical attributes recover 10 years after gap logging?
3) Do biophysical attributes recover 50 years after selective logging?

CHAPTER 3

Research Plan



Chapter 3

3.1 Design

Five different management prescriptions ('treatments') were required to address the two aims of this research. The first treatment provided baseline data to be used for comparison with data on forest disturbed by fire and on forest disturbed by logging. The alternative of using baseline data that has been previously collected in other studies in jarrah forest (e.g., the Kingston Study) was considered inappropriate because of the large spatial and temporal variation in structure and biotic composition that exists within the jarrah forest (outlined in Section 2.1). If data from other studies were used, differences between treatments could result from sampling biases or because of variation that naturally exists throughout the jarrah forest. The first treatment therefore consisted of forest that had never been logged, and had not been burnt for five years.

Understanding the effects of logging in jarrah forest is made complex because many different silvicultural practices have been used (outlined in Section 2.2). Contemporary silviculture uses four techniques (gap logging, shelterwood logging, thinning and selective logging). Each of these practices can be expected to have a different impact on biodiversity because of the varying intensity (represented by the numbers of trees that are cut down as well as ground disturbance) by which the forest is logged. Studies in other temperate forests show that the intensity of logging is an important determinant of how biodiversity is affected by logging disturbance (e.g., Kirkpatrick & Bowman 1982; Morrison 1992; Altegrim & Sjoberg 1995), with greater intensity usually having a greater impact on biodiversity (e.g., Huhta 1971; Spur ***et al.*** 1992;

Taylor *et al.* 1995; Summerville & Crist 2002; Moola & Vasseur 2004). My research assessed the impact of gap logging because it is the most severe of the contemporary practices (Abbott *et al.* 2003). Assessing the most severe practice was considered appropriate because it reveals the magnitude that biodiversity can be affected by contemporary practices and therefore sets a benchmark against which less intrusive procedures can be compared.

Two treatments of gap logging were investigated (representing the 2nd and 3rd treatment assessed in this research). The first focused on forest that had been recently gap logged and burnt. This treatment was required so that the immediate impact of logging disturbance could be compared with the immediate impact of fire. The second of the two treatments that assessed gap logging focused on forest that had been disturbed by gap logging more than seven years before. This treatment was selected to assess whether gap logging affects biodiversity over a longer period of time than fire does. These two treatments are therefore a chronosequence of forest impacted upon by gap logging disturbance.

The fourth treatment used in this research assessed the possibility that logging impacts on biodiversity for a longer time than seven years. Research in other temperate forests shows that forest biodiversity can be affected for several decades (e.g., Huhta 1971; McIver *et al.* 1992; Thompson *et al.* 1999; Moola & Vasseur 2004). Investigating forest disturbed by gap logging more than two decades previously cannot be carried out because this practice was officially implemented in 1985 (15 years before the commencement of this research). Prior to 1985, many different techniques were used to extract timber from the

jarrah forest (Stoneman ***et al.*** 2005). The one strategy that had been applied to most of the jarrah forest was selective logging, which is defined as cutting down trees that had economic value and leaving less valuable trees standing. Historians, and remaining stumps, show almost all the jarrah forest has been selectively logged by early foresters (Mills 1989). It was decided that the fourth treatment focus on forest that was selectively logged more than 20 years ago.

The fifth treatment assessed the impact of fire disturbance on unlogged jarrah forest. This treatment therefore represented the impact of natural disturbance on jarrah forest biodiversity. The same fire that was investigated in the second treatment (i.e., forest that had been gap logged and burnt) had to be investigated in this treatment because of the unique attributes of every fire (Grant & Loneragan 1999b; Burrows & Wardell-Johnson 2003). Disturbance by fire is influenced by factors that include climate and weather, topography, vegetation type, fuel amount and ignition sources (Gill 1975). If different fires were investigated for each treatment, then differences between unlogged and logged treatments could be the result of different properties of each fire (e.g., intensity, season, patchiness) (Wallace 1965; Peet & McCormick 1971; Christensen & Abbott 1989; Van Huerck ***et al.*** 1999; Burbidge 2003). Many of the early studies that investigated fire in jarrah forest have been criticised on methodological grounds (see review by Abbott & Christensen 1996). Although there have been many studies that have assessed the immediate impact of fire on different taxonomic groups in jarrah forest (e.g., Kimber 1974b; Springett 1976; McCaw 1988; Brennan 2002), no study has compared the effect of the same fire in logged and unlogged forest.

3.2 Assumptions of Retrospective Research

There are several assumptions that need to be recognised prior to undertaking retrospective research. The first is that all sites were similar prior to disturbance (Loyn 1993). The only way to address this assumption is to design an experiment that minimised the variation found between sites within each of the treatments (Underwood 1997). Given the natural variability that exists across small spatial scales in jarrah forest (Havel 1975a,b; Wardell-Johnson & Horwitz 1996) this is a difficult assumption to meet. It was decided that all study sites had to occur in a small region (<10 km^2) to reduce natural variability among sites. Within this region, treatments were represented by four sites to provide a measure of the natural variability that exists within this region, and to reduce the risk that the site(s) selected are not representative.

The second assumption in retrospective research is that all sites follow the same pattern or trajectory of recovery following disturbance (Majer & Nichols 1998). This problem is difficult to overcome because it is not known whether biotic communities follow the same trajectory after logging disturbances in the jarrah forest. In the only retrospective research that has addressed this assumption in jarrah forest, Brennan (2002) showed that spider composition was similar between sites disturbed by the same management prescription (i.e., forest either disturbed by fire 0, 3, 6 or 9 years ago, or forest disturbed by mining 0, 3, 6 and 9 years ago). In my research, this assumption was important to consider when comparing the two treatments that assessed past and present gap logging disturbance. I addressed this assumption by conducting pilot surveys prior to

undertaking research to ensure that all sites in these treatments had been perturbed by similar disturbance. This was achieved by counting the number of stumps per site, and ageing the stumps (see method by Abbott and Loneragan 1982), and by assessing earlier fire disturbance patterns (e.g., carbon residue of logs, and the height to which flames affected trees). If sites appeared to be disturbed in a similar manner, then it was assumed that biodiversity communities would follow the same trajectory after gap logging disturbance.

3.3 Study Area

Identification of potential study areas was carried out in consultation with a representative of Department of Conservation and Land Management (Pat Collins) using 1:25000 maps of the management history. Pilot surveys evaluated the feasibility of potential areas. The jarrah forest that surrounds Mt Dale was selected because it contained the range of management practices and fire histories that were required to address the study's aims, and it was easily accessible. Mt Dale is in the 'Jarrah Northwest' according to the delineations in the WA RFA (1999) (see Fig. 2.4, pg.18). This region has been heavily exploited for timber (Herbele 1997). Of 670 600 ha that existed prior to European arrival, 1.2 % is unlogged (calculated from data in Western Australian Regional Forest Agreement) (Commonwealth of Australia and Government of Western Australia 1999).

Mt Dale (32 ^{o} 06 'S, 116 ^{o} 17 'E) is 45 kilometres East - South - East (114 ^{o}) of Perth city (Fig. 3.1). It is an area of high relief (548 m above sea level) which, at the highest elevation, supports a woodland dominated by wandoo and jarrah. Surrounding the mountain is jarrah forest in Occidental and Dale forest blocks.

Small patches of unlogged forest exist in Dale forest block. Both forest blocks are dominated by jarrah interspersed with marri (*Corymbia calophylla*) and sheoak (*Allocasuarina fraseriana* (Miq.) L.A.S.Johnson). The subcanopy contains all three species together with *Banksia grandis* Willd., *Persoonia longifolia* R.Br. and *P. eliptica* R.Br. This tree community is typical of much of the northern jarrah forest (see Abbott & Loneragan 1986; Dell & Havel 1989).

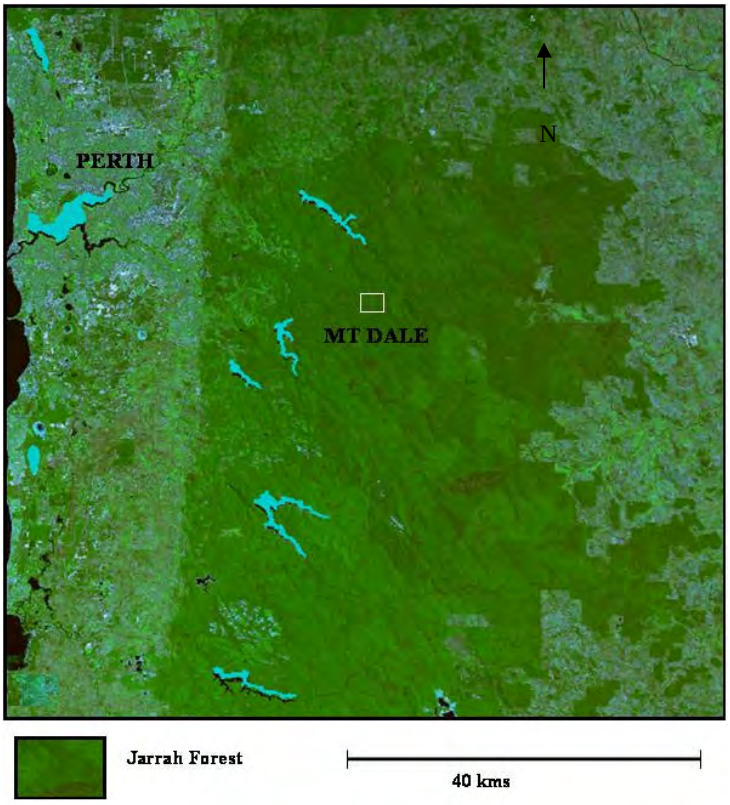


Fig. 3.1 The location of Mt Dale in the jarrah forest of south Western Australia.

Occidental and Dale Forest blocks were first logged before 1900 (Herbele 1997). Although principally managed for water conservation, Occidental Block and the south of Dale Block were selectively logged until 1940 (Fig. 3.2, Overlay 1). Between 1940 and 1990 there was little activity in the area. In 1990, much of the forest located north of Mt Dale road and northwest of Darking Ridge road in Occidental Block was harvested, primarily by gap logging (Overlay 2). In the

summer of 2000, the forest south of Mt Dale Road in Occidental block was harvested using gap logging and shelterwood techniques (Overlay 3). A moderate intensity regeneration burn (501-3000 kWm^{-1}) was conducted in Occidental forest block on the 5th of May 2000 between 1345-1530 (pers comm. Kevin Pollock, CALM Mundaring) (Overlay 4). The rate of spread was 60m-70m/hr in recently logged areas, and 35 m/hr in unlogged areas, and had a flame height between 0.7-1.5 m (pers comm. Kevin Pollock, CALM Mundaring). All other areas have been unburnt for at least five years. The five treatments were therefore able to be investigated in this region (Table 3.1).

Table 3.1 Description of the five disturbance regimes ('treatments') assessed in this research.

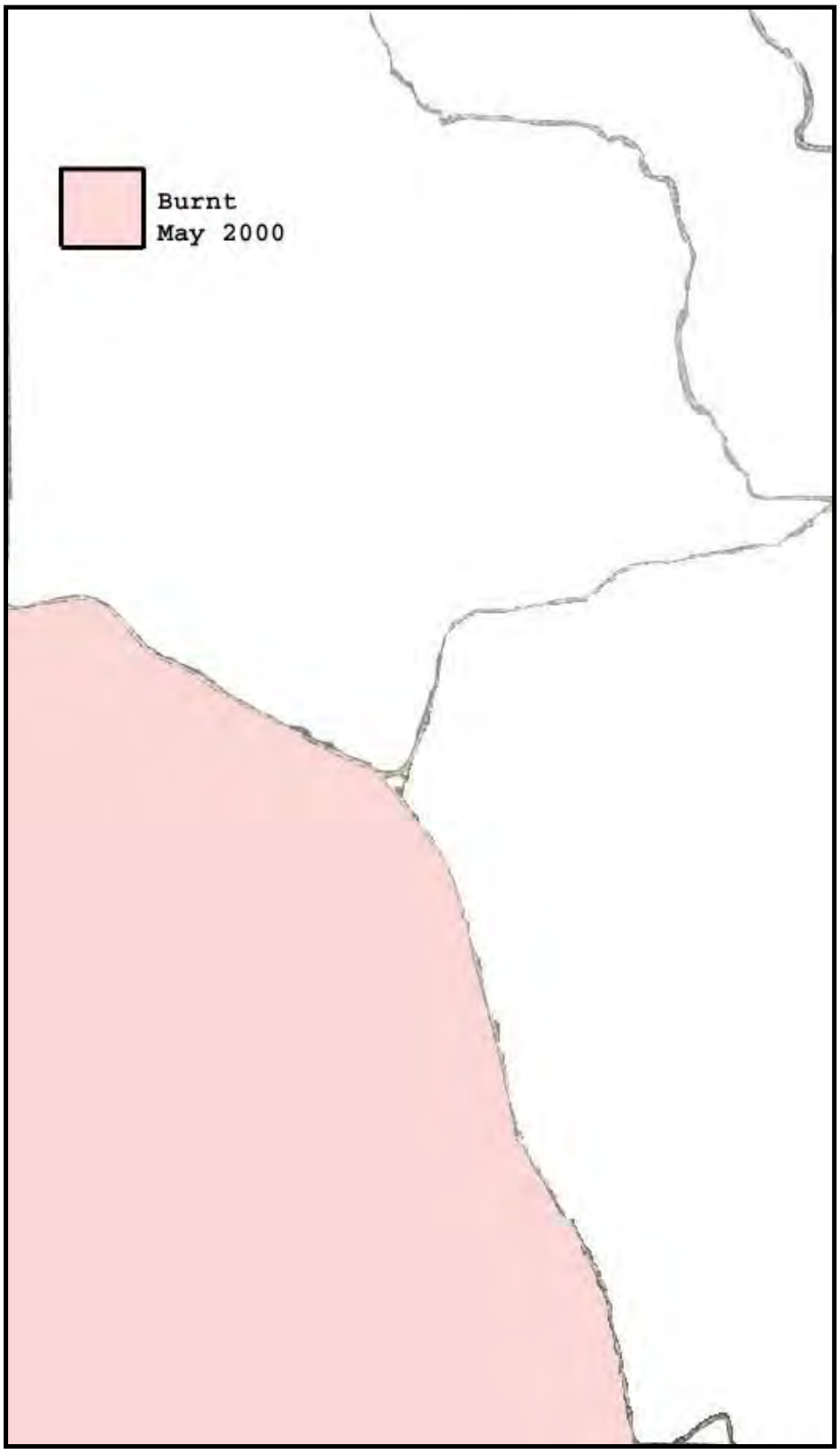
Logging History	Name of Treatments	Brief Description
Unlogged	'Undisturbed'	Unlogged forest that had not been burnt for at least five years.
	'Burnt 2000'	Unlogged forest burnt with a medium temperature fire in May 2000.
Logged	'Gap logged/ Burnt 2000'	Forest gap logged in the summer of 2000 and followed with a medium temperature fire in May 2000.
	'Gap logged 1990'	Forest gap logged in 1990, and not burnt for at least five years.
	'Forest selectively logged < 1950'	Forest selectively logged more than 50 years ago and not burnt for at least five years.

3.4 Study Sites

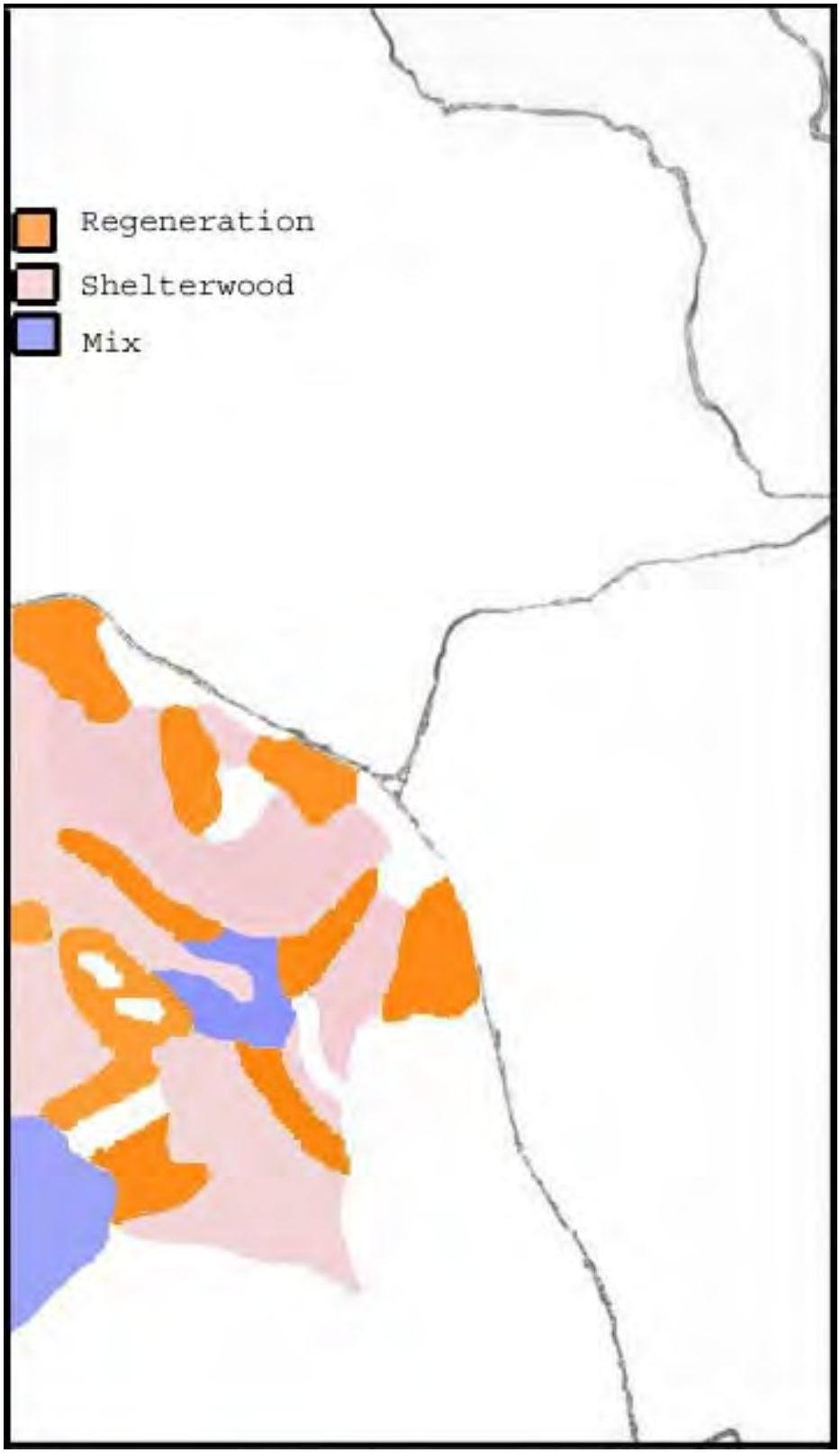
Each treatment was represented by four one-hectare sites (Fig. 3.2), located on slopes with a northerly aspect and on shallow gradients (mostly 1 – 5 ^{o}). Soils of this region are typically complex, having derived from various elements of a lateritic profile, and formed under peneplain conditions during the Oligocene (33.7 to 23.8 mya) and Miocene (23.8 to 5.3 mya) periods (Johnstone *et al.* 1973; Churchwood & Dimmock 1989). Typically, these soils are dominated by ferruginous gravel, lateritic boulders with sheet laterite, and mottled kaolinitic

clay (Abbott & Loneragan 1986). This overlies mottled yellow-brown clay subsoil (Beard 1990). To limit edaphic variation, forest on shallow to moderately deep gravely sandy soil over laterite duricrust was selected as sites. Sites were placed at least 30 metres from Mt Dale road (unsealed, limited access) and other limited-access roads.

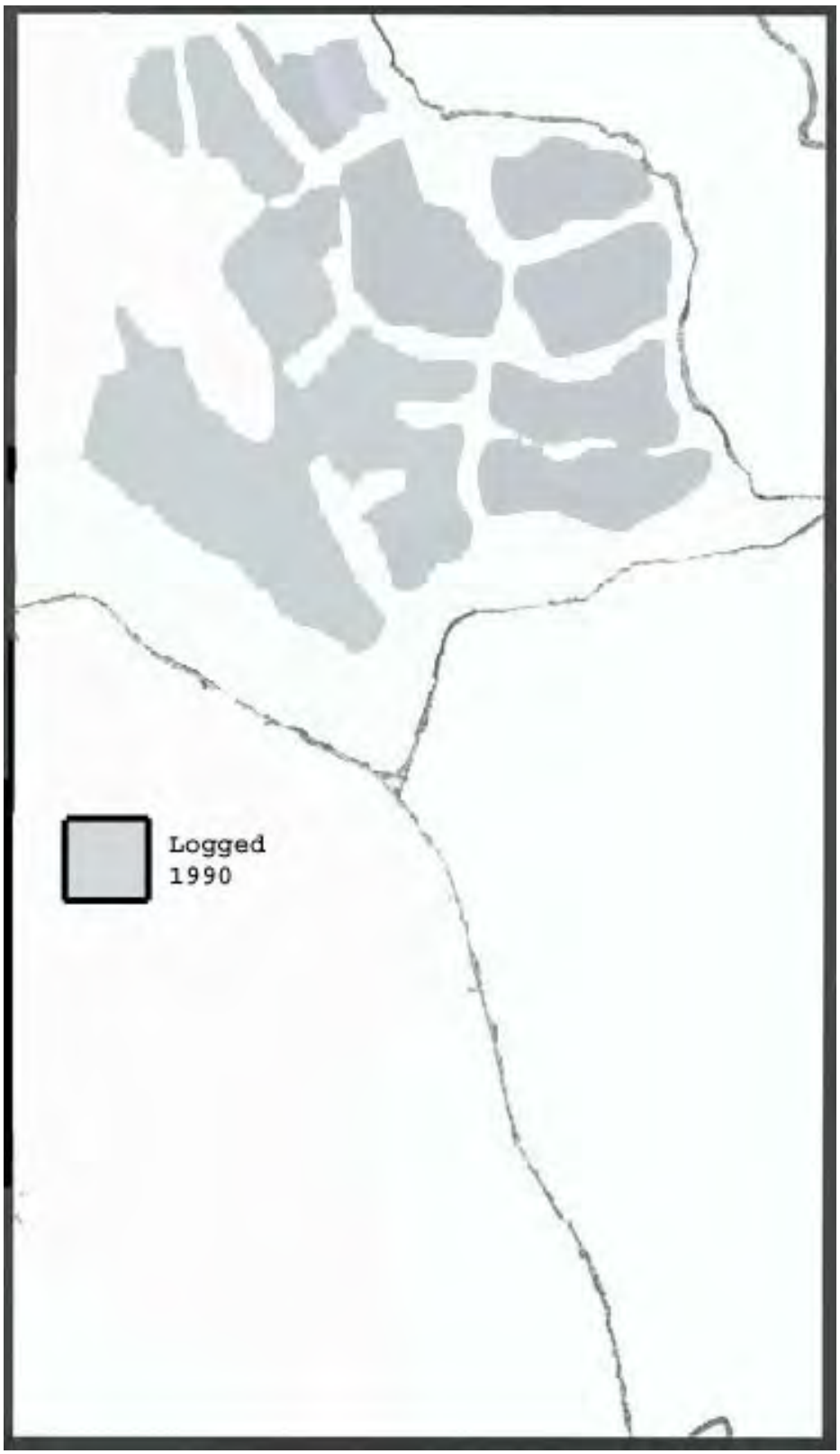
A Global Positioning System (GPS) was used to record the location of each site (Appendix 2). With exception of the 'Burnt 2000' treatment, all sites of the same treatment were at least 0.5 kilometres apart and in different logging coups (Fig. 3.2). Due to limited site availability, three of the burnt treatments were closer than 0.5 kilometres apart (see Fig. 3.2).



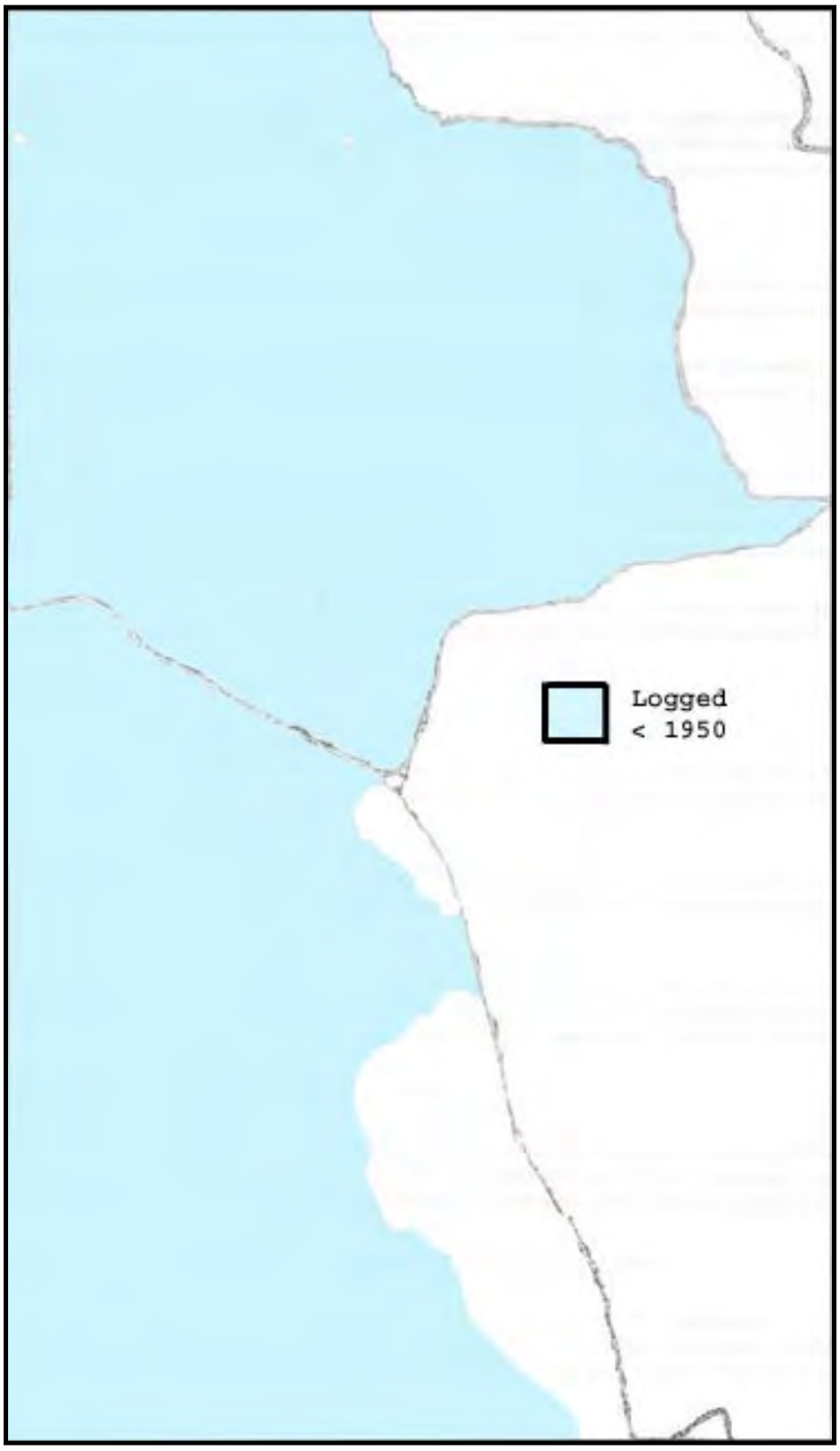
Overlay 4. Shaded area represents forest burnt in a regeneration burn conducted in May 2000.



Overlay 3. Shaded areas represent forest that was logged in 2000 by either i) regeneration release (gap logging), ii) shelterwood logging and iii) a mixture of shelterwood and regeneration release.



Overlay 2. Shaded area represents forest gap logged in 1990.



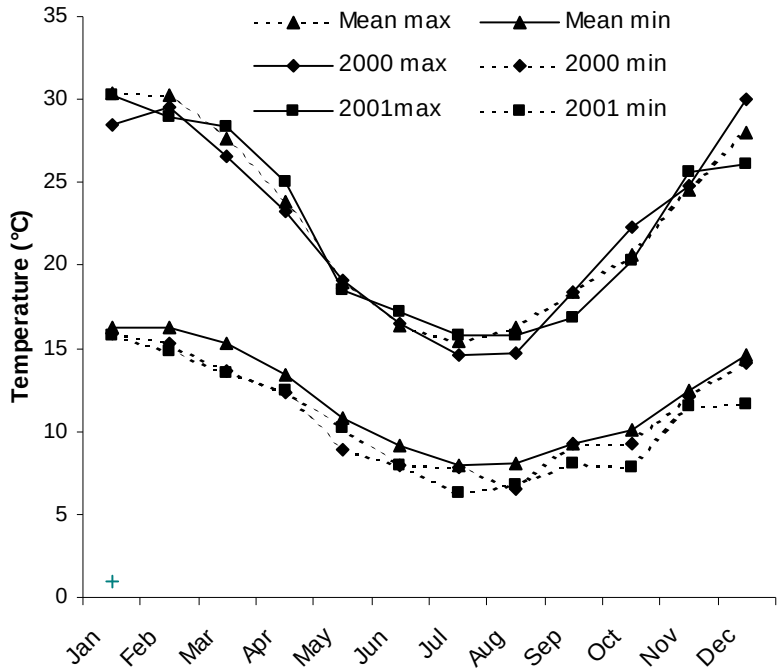
Overlay 1. Shaded area represents forest that was selectively logged prior to 1950.

Fig. 3.2 Aerial photo of region where research was undertaken (1:20000). White squares represent study sites. 'Un'-Undisturbed, 'B'- Burnt in 2000, 'GLB' - Gap Logged/ Burnt in 2000, 'GL 90' – Gap Logged in 1990, and 'SL'- Selectively Logged prior to 1950.

3.5 Weather During Study

The nearest weather station to the study area was at Kalamunda weather station (31^0 98 'S 116^0 15 'E). This weather station is located within the jarrah forest and is representative of the rainfall patterns at Mt Dale. Field research in this study began in July 2000 and ended in late 2001. Total rainfall for 2000 (1192 mm) was 11% greater than the average (1074mm), and for 2001 (860 mm) was 20% lower. Compared with mean monthly rainfall, the study sites had above average rainfall at the beginning of 2000, then became slightly below average until July 2000 (Fig. 3.3). After this, the typically 'Mediterranean' trend of wet winters (July-Aug), relatively dry springs (Sept - Nov) and dry summers (Dec - Feb) was followed. Less rainfall occurred in May 2001, but this was followed by a wet July and August. The mean daily maximum and minimum temperature ranges were similar to the average (Fig. 3.3). Mean monthly maximum temperature was similar to the long-term trends. Mean monthly minimum temperatures were 0.89 ° C (SD ± 0.28) lower than the average in 2000, and 1.39 ° C (SD ± 0.28) lower in 2001.

a) Temperature



b) Precipitation

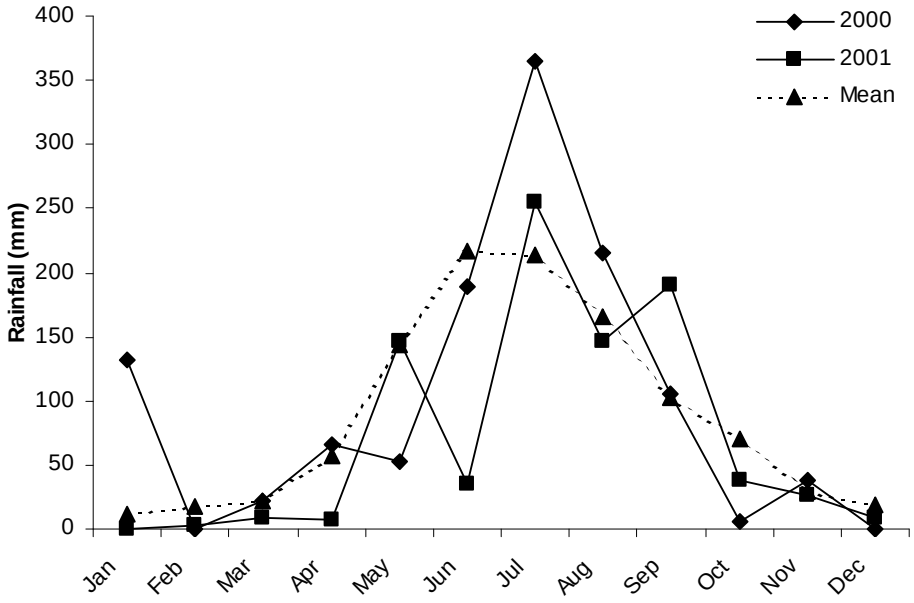


Fig. 3.3 The (a) mean monthly maximum and minimum temperature for 2000 and 2001 and the long term average (1908-1994), and (b) the total monthly precipitation recorded for the years 2000 and 2001, and the average rainfall for the period between 1908 and 1994. Information was provided by the Australian Bureau of Meteorology.

3.6 Measuring Biodiversity in Jarrah Forest

Biodiversity is difficult to quantify because it embodies the totality of life, and the variability within and between the various organisational levels (Department of Conservation and Land Management 2004b). Biodiversity is often considered at three levels (Australian Department of Environment, Sport and Territories 1996; Swingland 2000):

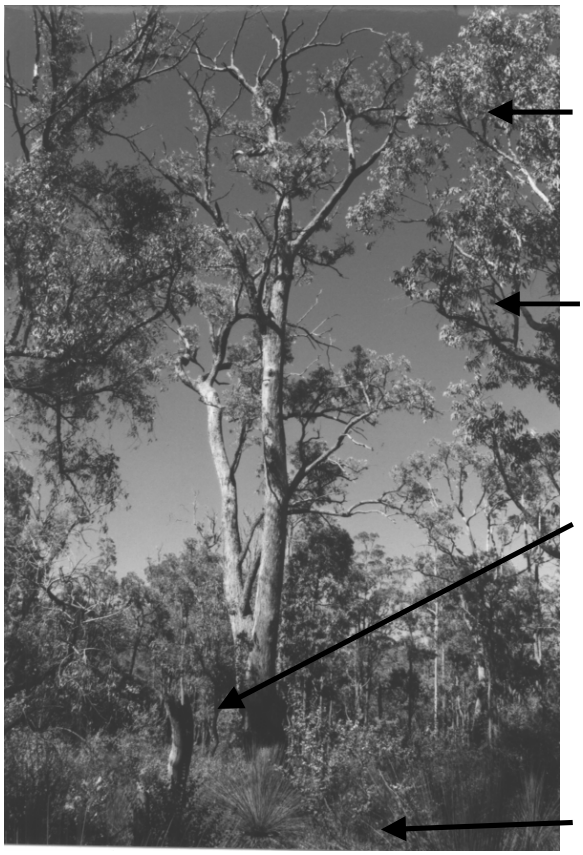
- ***Genetic diversity.*** The variety of genetic information contained in all life forms. Genetic diversity occurs within and between the populations of organisms that comprise individual species as well as among species;
- ***Species diversity***. The variety of species on earth.
- ***Ecosystem diversity.*** The variety of habitats, biotic communities and ecological processes.

In jarrah forest, like most ecosystems on Earth, the vast majority of species are not described, their genetic variability is unknown, and their roles in ecological processes are hypothesised. Given the breadth of the biodiversity concept, surrogates were required to assess the biodiversity of Mt Dale. Many different surrogates have been used to measure forest biodiversity, ranging from ecosystem diversity through to genetic diversity (Margules 1996; Stork *et al.* 1997; Catteral *et al.* 2004). The choice of surrogate depends on the level of organisation that is being assessed (Noss 1990). For example, if understanding the biodiversity at a regional scale is required, the identity and distribution of particular habitat types could be mapped as a surrogate of biodiversity (e.g., design of the C.A.R. reserve system in jarrah forest). In contrast, the ongoing viability of different populations is often most accurately assessed by measuring genetic diversity (e.g., Van Rossum *et al.* 2002; Brouat *et al.* 2003) or breeding

success (Bourque & Villard 2001; Duguay *et al.* 2001). In this research, I assessed biodiversity at the 'community-ecosystem' scale (*sensu* Noss 1990). This spatial scale represents a relatively homogeneous landscape with respect to community composition and abiotic aspects. Stand structural diversity and species diversity are two useful surrogates for measuring biodiversity at this spatial scale (e.g., Spence *et al.* 1997; Lindenmayer *et al.* 2000; Canterbury *et al.* 2000).

3.6.1 Stand Structural Diversity

Stand structural diversity is a useful measure of biodiversity (Boncina 2000; Rolstad *et al.* 2002; Tews *et al.* 2004). Since MacArthur & MacArthur's (1961) seminal research, measuring attributes such as the volume and decay of coarse woody debris, the vertical cover of the understorey at different heights, and the amount of leaf litter provides useful information about the organisms that use these resources (Fig. 3.4). Measuring how these, and other, structural attributes are affected by logging and fire helps identify the types of organisms that may be affected by these disturbances (Lindenmayer *et al.* 2000). Chapter 4 describes how fire and logging affects structural attributes at Mt Dale.



Vertical and horizontal canopy cover influences microclimates including solar radiation and moisture regimes, and provides direct foraging, nesting or roosting resources for many species.	Recher & Holmes 1985, Raegan 1992, Atlegrim & Sjoberg 1995, McCune & Amsberry 1997.
Tree composition, size and decay influences nutrient cycling and can impact on foraging, nesting or roosting resources for many species.	Berg *et al*. 1994, Kuusinen 1994, Lindenmayer & Franklin 1997a,b.
The volume and decay of coarse woody debris (CWD) affects nutrient cycling, and provides foraging, nesting or roosting resources for many species. CWD also provides favourable germination and establishment of some plants, bryophytes and fungi.	Soderstrom 1988, Lattin & Moldenke 1992, Siitonen 1994, Bader *et al.* 1995, Lindblad 1998, Grove *et al.* 2002.
Litter depth, litter complexity and litter cover influences soil moisture, temperature and plant productivity, and provide foraging, nesting or roosting resources for many species.	Facelli & Pickett 1991, Xiong & Nilsson 1999, Hooper *et al.* 2000, Judd 2004.

Fig. 3.4 Some examples of structural attributes that can influence forest biodiversity.

3.6.2 Species Diversity

Species diversity is internationally recognised as a measure of forest biodiversity (Hunter 1990; Oliver *et al.* 1998). In Western Australia, the Wildlife Conservation Act (1950)[1] focuses on protecting what is now recognised as 'biodiversity' by ensuring the maintenance species diversity. There are many ways that 'species diversity' can be measured (see review by Burton *et al.* 1992; Whittaker 2002). In this study, two measures have been used. The first was to measure alpha (α) diversity which refers to the number of species within a site (coined 'species richness') (Whittaker 1972). This measure was selected because managers of forests in Western Australia have attempted to conserve species

diversity by maintaining the same number of species after a particular disturbance as occurred prior to it (e.g., Gioia & Piggot 2000; Abbott *et al.* 2002; Bradshaw 2002). This has meant that research has focused on species richness as the way to measure species diversity (e.g., Abbott & Van Huerck 1985; Norwood *et al.* 1995; Burrows *et al.* 2002). The second way species diversity was measured at Mt Dale was by assessing beta (β) diversity. β diversity is the change in species assemblages that occurs among sites (Whittaker 1972). This second measure of diversity was selected because changes in the assemblages of species can occur following disturbance yet species numbers can remain equivalent (e.g., Huhta 1971; Buddle *et al.* 2000; Willett 2001). Both α and β diversity were measured to ensure that any changes in species diversity following logging or fire were identified.

Assessing α and β diversity is particularly useful when different species of the same taxa are placed within different guilds (Simberloff & Dayan 1991; Hansen *et al.* 1995; Summerville & Crist 2002; Decocq *et al.* 2004). A guild is defined as a group of species that have similar resource requirements (Root 1967; Blondel 2003). This technique is useful because a change in the diversity of species within a guild after disturbance can highlight how specific resources have been affected. Furthermore, guilds can help identify other taxa that may be detrimentally affected by a disturbance because they utilise similar resources (e.g., Passos and Oliveira 2002 ; Tejeda-Cruz & Sutherland 2005). Therefore, species within each taxa were placed in guilds that represent resources that could be affected by logging.

[1] The Wildlife Conservation Act (1950) is the principal legislation directly relating to biodiversity conservation. It is currently being replaced with the Biodiversity Conservation Act.

3.6.3 Selecting Appropriate Taxa

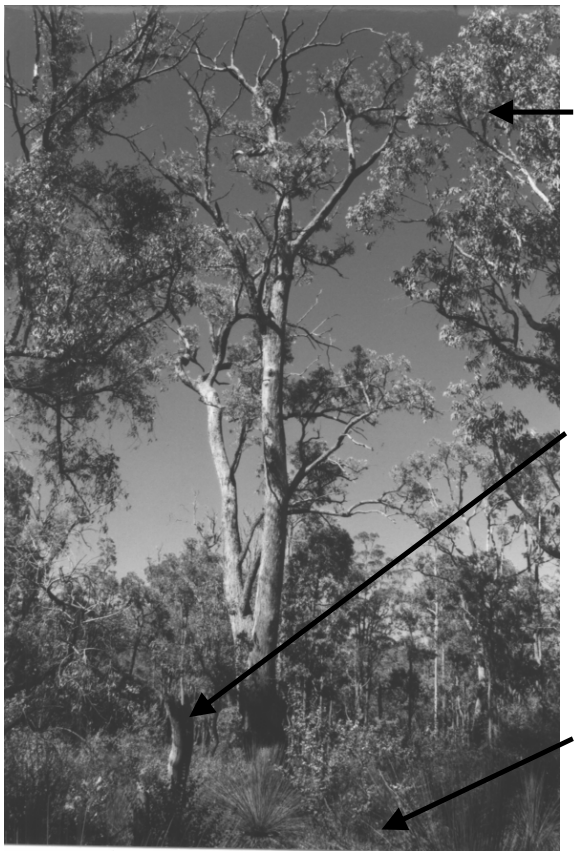
Many taxa were considered when this research was formulated. Six criteria were used to select appropriate taxa:

1) Taxa had to have a substantial functional role in the ecosystem and respond to environmental changes in a quantitative, predicable way (Noss 1990),
2) Taxa had to have life history characteristics that are likely to respond to logging and fire disturbance at the spatial scale that is being investigated (Noss 1990; Kremen 1992; Dale & Beyeler 2001),
3) Taxa had to be ecologically and taxonomically diverse (Medellin *et al.* 2000),
4) Taxa had to be relatively easy to sample (Noss 1990; Dale & Beyeler 2001),
5) The methods for sampling and identifying the taxa had to be cost-effective (Noss 1990), and,
6) There had to be extensive available taxonomic knowledge about the taxa to avoid cost and time seeking background information (Noss 1990).

Three taxa were selected after feasibility studies were undertaken. Understorey plants, ground dwelling spiders and birds were considered appropriate taxa to use because they meet the six criteria, and because these taxa have been successfully used in previous research that assessed the immediate impacts of logging in jarrah forest (see Abbott *et al.* 2002; Burrows *et al.* 2002b; Strehlow *et al.* 2002; Abbott *et al.* 2003).

Indicator 1: Understorey Plants

'Understorey' plant communities are defined in this research as all vascular plant species that are less than 5m in height. In other ecosystems, they have been shown to be useful indicators in assessments of logging and fire (e.g., Gashwiler 1970; Loyn *et al.* 1983; Dickinson & Kirkpatrick 1987; Hickey 1994; Halpern & Spies 1995; de Grandpre *et al.* 2000; Thysell & Carey 2000). Understorey plants are considered 'keystone' within forest communities because as autotrophic life forms, they directly and indirectly provide much of the food required by forest organisms and their structure is utilised by many organisms for reproduction and shelter (Halpern & Spies 1995). There are also many examples of plant-abiotic, plant-plant, plant-animal, and plant-microbe interactions that have major effects on productivity, population sizes, species composition and ecosystem function (Lamont 1992). Changes in fire regimes, canopy cover, coarse woody debris, soil properties and leaf litter that result from logging are likely to impact on the diversity of understorey communities in jarrah forest (Fig. 3.5). Previous studies have shown that plant diversity responds to changes caused by fire and logging disturbance (e.g., Christensen & Kimber 1975; Bell & Koch 1980; Dixon *et al.* 1995; Burrows *et al.* 2002b; Burrows & Wardell-Johnson 2003). Therefore, the diversity of understorey plants was used to assess impacts associated with logging and fire disturbance at Mt Dale



Canopy cover influences understorey diversity by moderating microclimates including solar radiation and moisture regimes.	Ganzhorn *et al.* 1990, Thiollay 1992, Cannon *et al.* 1994, Breshears *et al.* 1998, Traut & Muir 2000.
Coarse woody debris can physically inhibit plant growth. Furthermore the decomposition of CWD has been shown to be an important site for germination for some species.	Sollins *et al.* 1981, Harmon & Franklin 1989, McKenny & Kirkpatrick 1999, Grove *et al.* 2002
Litter depth, litter complexity and litter cover can influence understorey plant diversity by influencing soil moisture, temperature and productivity.	Barret 1931, Wallace & Hatch 1952, Facelli & Pickett 1991, Xiong & Nilsson 1999, Burrows *et al.* 2002b.

Fig. 3.5 Structural attributes that could be affected by logging or fire, and may influence understorey plant diversity at Mt Dale. References represent exemplars in other forests where specific changes in forest structure have affected the diversity of understorey plants.

Indicator 2: Ground dwelling Spiders

Spiders are useful surrogates of biodiversity (Churchill 1997). As major predators of other arthropods, the presence or absence of particular species provides a strong indication of forest habitat quality (McIver *et al.* 1990; Hopper *et al.* 1996). Furthermore, as some of the oldest and most persistent of terrestrial organisms in Western Australian forests, many spider families have evolved to exploit specific environmental niche that are deleteriously impacted by logging and fire (Main 2001). Previous studies in jarrah forest have shown that the diversity of ground dwelling spiders (hereafter referred to as 'ground spiders')

can be used to evaluate the effects of mining and fire disturbance (e.g., Nichols & Burrows 1985; Mawson 1986; Brennan 2002). Changes to canopy cover, ground vegetation cover, leaf litter properties and coarse woody debris following logging or fire is expected to influence the types of ground spiders that exist within disturbed ecosystems (Curry *et al.* 1985; McIver *et al.* 1992; Koponen 1995; Willett 2001; Fig. 3.6). Therefore, the diversity of ground spiders was used to assess impacts associated with logging and fire disturbance at Mt Dale.

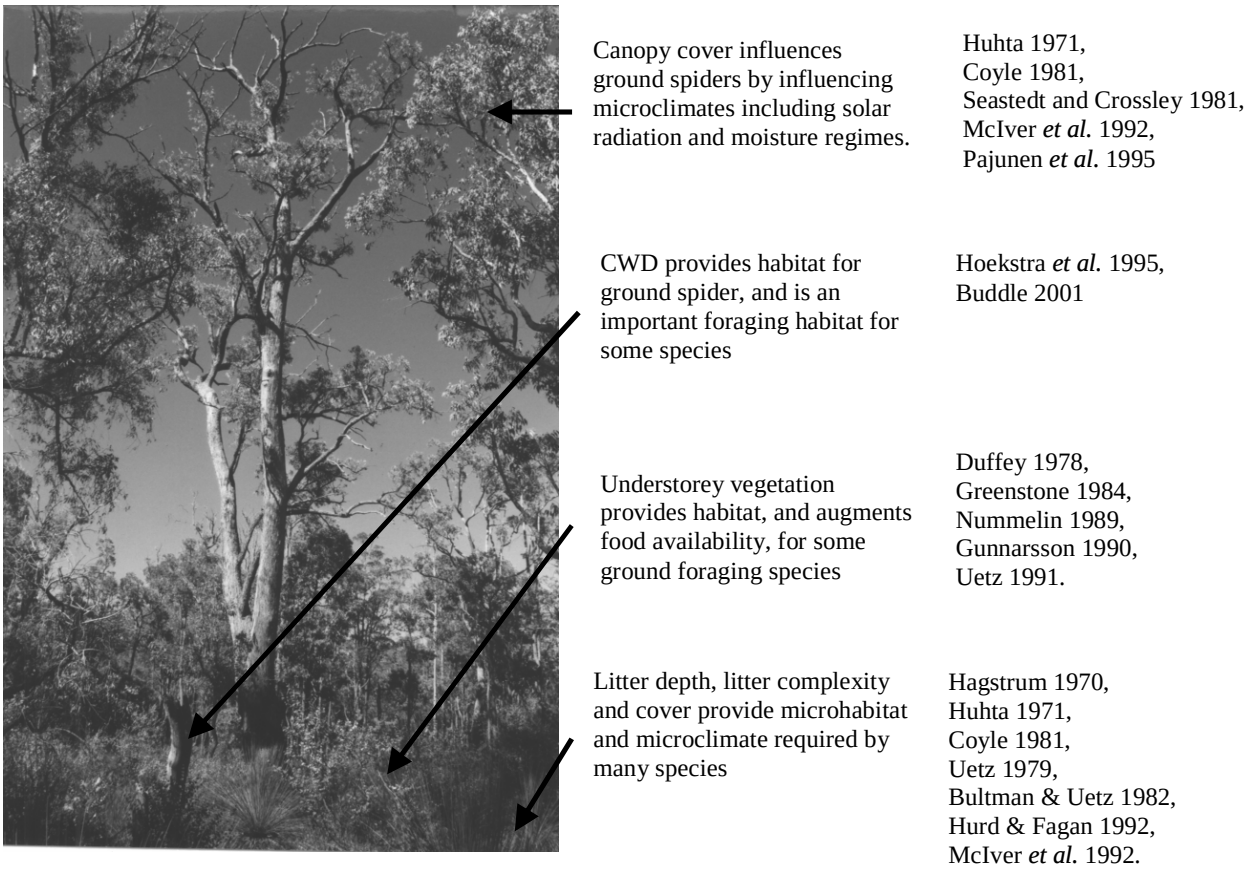


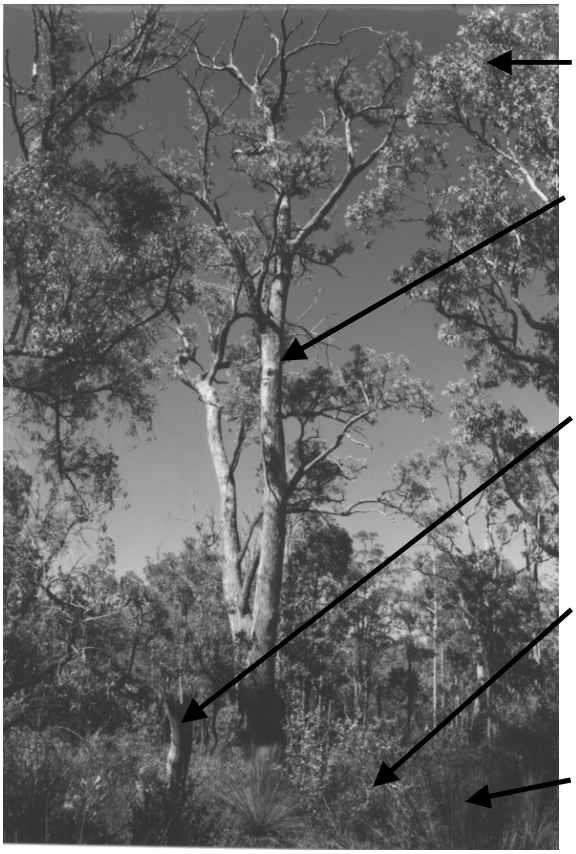
Fig. 3.6 Structural attributes that could be affected by logging or fire, and may influence ground spider diversity at Mt Dale. References represent exemplars in other forests where specific changes in forest structure have affected the diversity of ground spiders.

Indicator 3: Birds

Extensive taxonomic knowledge, conspicuous behavior and predictable responses to disturbance make birds the most used taxa for testing ecological

theory and monitoring environmental change in terrestrial environments (Recher 1988; Temple & Wiens 1989; Hansen *et al.* 1993; Robichaud *et al.* 2002). In Australia, birds have long been used in assessments of management practices when exploitation became a contentious issue (e.g., Loyn 1979; Recher *et al.* 1980; Christensen 1982; Smith 1985a,b; Loyn & MacFarlane 1984; Loyn 1993; 1985a). These studies have allowed the results of continuing logging practices on bird communities to become foreseeable. They have subsequently influenced how forests are managed in many regions. For example, maintaining a natural mix of tree species, and protecting some 'habitat' (hollow bearing) trees and old logs within logged stands are thought to enhance the recovery of forest specialists in production forests (e.g., Loyn 1985b; Recher 1991; Kutt 1994; Loyn 2000). Furthermore, protection of riparian habitat (e.g., Loyn 1979; Smith 1985b), interspersing coupes with floristic and structurally diverse forest (Recher *et al.* 1987) and longer rotations (Loyn 2000), are all important conservation measures now considered by forest managers.

Birds are relatively depauperate in species richness and density in jarrah forest compared with forests in eastern Australia (Abbott 1999). In spite of this, birds have been used successfully to assess the impact of various disturbances, including logging (Abbott & Van Huerck 1985a; Norwood *et al.* 1995; Craig 1999), mining (Collins *et al.* 1985; Wykes 1985; Curry & Nichols 1986; Armstrong & Nichols 2000), and fire (Kimber 1974b; Christensen & Kimber 1975; Christensen *et al.* 1985). These studies, and many others, show birds require specific structural attributes to forage and nest that are likely to be impacted upon by logging (Fig. 3.7). Therefore, the diversity of birds was used to assess impacts associated with logging and fire disturbance at Mt Dale.

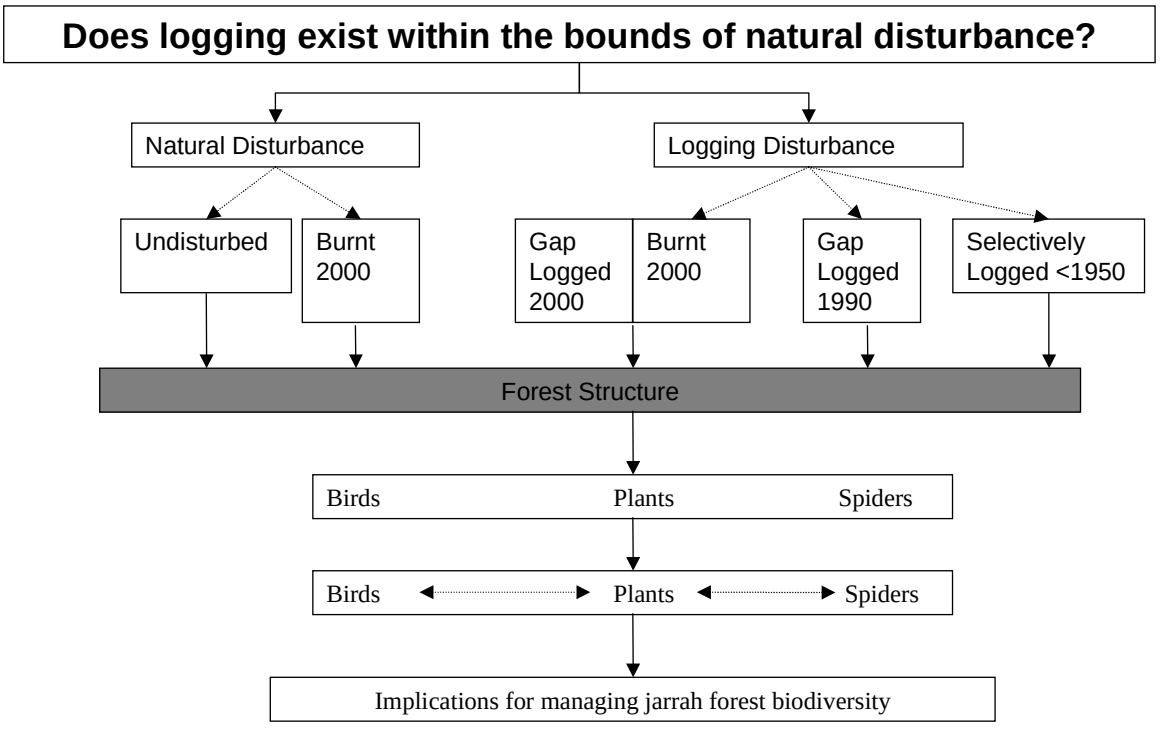


Canopy cover provides foraging, nesting and roosting habitat	Spur *et al.* 1992, Peck 1989 Taylor & Haseler *et al.* 1993, Hansen *et al.* 1995
Old trees provide foraging, nesting and roosting habitat	Smith 1985a,b, Recher 1991, Kutt 1994, Virkala *et al.* 1994, Kavanagh & Bamkin 1995.
Coarse woody debris are important foraging habitat for some species.	Shiemske & Broka 1981, Loyn & MacFarlane 1984, Laven & MacNally 1998, Tomialojc & Wesolowski 2004.
Understorey plants provide foraging, nesting and roosting habitat.	Wong 1986, Thiollay 1992, Bersier & Meyer 1994 Aleixo 1999.
Litter leaf houses arthropods that are important foraging habitat	Medin & Booth 1989, Robinson & Robinson 1999.

Fig. 3.7 Structural attributes that could be affected by logging or fire, and may influence bird diversity at Mt Dale. References represent exemplars in other forests where specific changes in forest structure have affected the diversity of birds.

CHAPTER 4

Forest Structure



Chapter 4

4.1 Introduction

The term 'forest structure' describes the physical arrangement of attributes that provide three-dimensional structure (Spies 1998; Stone & Porter 1998). Forest structure comprises, among other measurements, the distribution and abundance of plant species, their vertical and horizontal spatial patterns, size, age, and decay. The spatial and temporal arrangement of structural attributes influences the persistence and composition of biota within forests (MacArthur & MacArthur 1961; Recher 1969, 1971; Heliovaara & Vaisanen 1984; Adams & Morrison 1993; Siitonen & Martikainen 1994; Spies 1998; Coops & Catling 2000). Changes to structural attributes caused by logging have been implicated in the loss of forest dependent species (e.g., Heliovaara & Vaisanen 1984; Siitonen 1994). Consequently, the recognition and protection of structural attributes has become a critical part of conservation strategies designed to comprehensively conserve biodiversity within production forests (Lindenmayer & Franklin 1997b; Palik & Engstrom 1999).

No one set of attributes is used to describe the structure of a forest. Instead, a variety of attributes are measured in accordance with the objectives of the research. For example, silviculture in jarrah forest requires the measurement of size and density of the commercially important species (primarily jarrah), the potential of subdominant jarrah to grow into millable timber, and the cover of jarrah ground coppice prior to logging (see Department of Conservation and Land Management 1995). These guidelines have resulted from studies that have assessed the growth rate and regeneration of jarrah following logging (e.g., Chandler 1935; Stoate & Bednall 1940; Harris 1953; Abbott & Loneragan 1982),

and how growth rate and regeneration is affected by thinning (e.g., Kimber 1965; Kimber 1967; Abbott & Loneragan 1983a; Stoneman 1986; Stoneman and Whitford 1995), fire (e.g., Peet 1965; Schuster 1980; Abbott & Loneragan 1983b), site quality (e.g., Abbott & Loneragan 1983c), fertiliser (e.g., Stoneman and Whitford 1995; O'Connell & Mendham 2004) and tree characteristics (e.g., Stoneman *et al.* 1995; Stoneman and Whitford 1995). Other structural attributes have been measured to investigate previous logging regimes (Abbott 1984b), water relations (e.g., Greenwood *et al.* 1985; Stoneman & Schofield 1989; Stoneman *et al.* 1989a; Crombie 1992; Crombie 1997) and habitats required by individual species (e.g., Faunt 1992; Williams & Faunt 1997; Whitford 2001; Whitford 2002; Whitford and Williams 2002) (Fig. 4.1).

Fig. 4.1 Some attributes of 'forest structure' that are measured to investigate fire, dieback, water relations and fauna habitat in jarrah forest.

An important factor that influences forest structure is disturbance (Spies 1998; Seymour & Hunter 1999; Boncina 2000). Fire, earthquakes, floods, blowdowns

and tree-fall destroy, transform, transport and create attributes of forest structure (White & Pickett 1985; Attiwill 1994; Namikawa *et al.* 1997). As outlined in Chapter 1, fire is the most frequent and widespread 'natural' disturbance in jarrah forest. The impact fire has on structural attributes depends primarily on its intensity. This in turn depends on many factors that include the frequency of past fires, the fuel load and climatic conditions prior to and during the fire (e.g., temperature and rainfall). Fires that are considered to have 'very high' intensity (defined as > 7000kWm^{-1} by Cheney 1981) can remove almost all vegetation, and fine woody debris, and could kill mature jarrah trees (Peet & Williamson 1968; Abbott & Loneragan 1983). In contrast, 'moderate' intensity fires (defined as 501-3000 kWm^{-1} by Cheney 1981) cause reductions in the volume and cover of leaf litter, fine woody debris and coarse woody debris, but might not impact on tree structure (Christensen & Abbott 1989).

Logging has profoundly influenced many structural attributes in jarrah forest. Although selective logging in Mundlimup and Amphion forest blocks (*c.* 25 km south of Mt Dale) did not affect the spatial pattern and segregation of five common species (*Eucalyptus marginata, Corymbia calophylla, Banksia grandis, Allocausarina fraseriana* and *Persoonia longifolia*) (Abbott 1984b), early selective logging left a legacy of a younger, more open forest (Abbott & Loneragan 1986; Calver & Wardell-Johnson 2004). Early logging practices also removed the largest trees ('King Jarrah') (Abbott & Loneragan 1986), reduced the density of understorey plants such as *Persoonia longifolia* (Abbott 1984b), and increased the volume of woody debris (Mills 1989; Havel 1989b). Given the intensity of logging (number of trees logged per ha) has increased as a result of contemporary logging practices (see Section 2.3), it is predicted that current

management prescriptions will have a greater impact on forest structure than past selective practices. Furthermore, as studies in other Australian eucalypt forests have shown that fire affects different attributes of forest structure than logging (e.g., Kirkpatrick & Bowman 1982; Attiwill 1994; Lindenmayer 1995), it is also predicted that there will be large structural differences in forest disturbed by logging compared to forest disturbed by fire. These predictions will be assessed by addressing three questions:

1) Are there differences in forest structure in forest recently disturbed by fire compared with forest recently disturbed by gap logging and fire?
2) Does forest structure recover 10 years after gap logging?
3) Does forest structure recover 50 years after selective logging?

4.2 Methods

4.2.1 Selection of Structural Attributes

Structural attributes were measured at the stand scale (refered to as the 'patch' scale in jarrah forest silviculture, see Section 2.3) because it is at this scale that logging disturbance is managed in jarrah forest. Sixty-seven attributes were measured (Table 4.1). These attributes were selected because they provide important resources to organisms that are dependent on forests to exist, and because they are affected by logging in other temperate forests. Attributes were placed within one of five broad categories:

1) ***Composition and Decay of Trees***

The composition and age-distribution of trees within a stand influences the diversity of many taxa including birds, mammals, beetles and fungi (e.g., Berg *et al.* 1994; Kuusinen 1994; Recher 1996; Lindenmayer & Franklin 1997a,b; Palik & Engstrom 1999). Logging can alter both the composition and age-distribution of trees within a forest because specific species and size classes are selectively removed from the ecosystem (e.g., Six Dijkstra *et al.* 1985; Berg & Clement 1992; Namikawa *et al.* 1997; Ziegler 2000; Sullivan *et al.* 2001; Pregistzer & Euskirchen 2004). In jarrah forest, changes in the composition of trees as a result of logging were noticed in the early 1900s (Abbott & Loneragan 1986). In many stands, marri became more common after the selective removal of jarrah trees (Abbott & Loneragan 1986). Changes in the age structure of jarrah stands following logging has also been recognised (Abbott & Loneragan 1986; Calver & Wardell-Johnson 2004), leading to predictions that gap logging will create even-aged stands (Burrows *et al.* 2002b). Fifteen attributes were measured to assess the composition and decay of trees across sites at Mt Dale (Table 4.1).

Table. 4.1 The structural attributes measured at Mt Dale.

Tree Composition and Decay (Stand Basal Area $m^2 ha^{-1}$)	**Coarse Woody Debris** (Volume $m^3 ha^{-1}$)	**Tree** ($m^2 ha^{-1}$)* **and Log Size** ($m^3 ha^{-1}$)**	**Canopy Cover** (%)	**Ground Cover** (%)
1. Total	1. Total	1. Small Jarrah Logs	1. Total	1. Shrub 0-0.2m hgt
2. Total Codominant	2. Total Logs	2. Medium Jarrah Logs	2. Total Dominant	2. Shrub 0.2-0.4m hgt
3. Jarrah Codominant	3. Banksia Logs	3. Large Jarrah Logs	3. Jarrah Dominant	3. Shrub 0.4-1m hgt
4. Marri Codominant	4. Sheaok Logs	4. Small Non-jarrah Logs	4. Marri Dominant	4. Shrub 1-2m hgt
5. Total Dominant	5. Marri Logs	5. Medium Non-jarrah Logs	5. Total Codominant	5. Shrub Structural Diversity
6. Jarrah Dominant	6. Balga Logs	6. Large Non-jarrah Logs	6. Jarrah Codominant	6. Leaf Litter
7. Marri Dominant	7. Jarrah Logs	7. Small Jarrah Trees	7. Marri Codominant	7. Exposed Laterite
8. Total Subdominant	8. Jarrah Decay 1 Logs	8. Medium Jarrah Trees	8. Sheoak Codominant	8. Woody Debris
9. Jarrah Subdominant	9. Jarrah Decay 2 Logs	9. Large Jarrah Trees	9. Total Subdominant	9. All Stumps
10. Marri Subdominant	10. Jarrah Decay 3 Logs	10. Small Non-jarrah Trees	10. Jarrah Subdominant	10. Bare Soil
11. Jarrah Decay '1'	11. Jarrah Decay 4 Logs	11. Medium Non-jarrah Trees	11. Marri Subdominant	11. Other
12. Jarrah Decay '2'	12. Jarrah Decay 5 Logs	12. Large Non-jarrah Trees	12. Banksia Subdominant	
13. Jarrah Decay '3'	13. Standing Dead Trees		13. Sheoak Subdominant	
14. Jarrah Decay '4'	14. Dead Stumps		14. Other Subdominant Sp.	
15. Total Sheoak			15. Canopy Structural Diversity	

* small = (0 - 0.1 $m^2 ha^{-1}$), medium = (0.1 - 1 $m^2 ha^{-1}$), large = (> 1 $m^2 ha^{-1}$)
** small = (0 - 0.1 $m^3 ha^{-1}$), medium = (0.1 - 1 $m^3 ha^{-1}$), large = (> 1 $m^3 ha^{-1}$)

2) *Coarse Woody Debris*

The quantity and distribution of coarse woody debris (CWD) in different states of decay can influence the diversity of many taxa (e.g., saprotrophic fungi, lignicolous cryptogams, arthropods, amphibians, mammals and birds) (e.g., Lattin & Moldenke 1992; Siitonen 1994; Dupuis *et al.* 1995; Laven & MacNally 1998; Lindblad 1998; Hong *et al.* 1999; McKenny & Kirkpatrick 1999; Pyle & Brown 1999; Andrew *et al.* 2000; Butts & McComb 2000; Bowman *et al.* 2000; Grove 2000; MacNally 2001). The volume of CWD is significantly impacted by logging in other temperate forests (e.g., Bader *et al.* 1995; Bengtsson *et al.* 1997; Goodburn & Lorimer 1999; Siitonen *et al.* 2000; Ziegler 2000; Tomialojc & Wesolowski 2004). In jarrah forest, no study has quantified how coarse woody debris is impacted by logging, although the CWD that result from logging are predicted to influence fire regimes (Hobbs 2003) and nutrient cycling (Brown *et al.* 1996).

3) *Size Distribution of Trees and Logs*

The size distribution of trees and logs directly (e.g., providing habitat), and indirectly (e.g., through shade or the production of woody debris) affect many resources required by organisms that occur in forests (Hoiland & Bendiksen 1996; Siitonen 2001; McGee *et al.* 1999; Grove *et al.* 2002). For example, the occupation of hollows by mammals and birds is directly related to the size of logs and trees in many forests including the jarrah forest (Williams & Faunt 1997; Whitford & Williams 2001; Whitford 2002). As previously outlined, selective removal of particular size classes (e.g., large trees) means that logging can alter the distribution of trees of different sizes throughout the forest (Spies 1998). Studies in other temperate forests have shown the removal of large trees through

logging can result in large changes in the volume of coarse woody debris in the ecosystem, which in turn affects the diversity of organisms that require coarse woody debris as well as nutrient cycling (e.g., Gore & Patterson 1986; Huston 1996). For example, after repeated disturbance by logging, forests can come to consist of smaller trees which provide different resources to forest species which can decrease the volume of large, well-decayed, coarse woody debris (McGee *et al.* 1999; Grove *et al.* 2002). Twelve measures were used to asses the size distributions of trees and logs (Table 4.1).

4) ***Canopy Cover***

Canopy cover can be significantly reduced following logging (Franzeb 1983; Smale *et al.* 1985; Ganzhorn *et al.* 1990; Morrison 1992; Cannon *et al.* 1994; Fredericksen & Mostacedo 2000). Changes to the horizontal and vertical cover provided by this vegetation have been related to the diversity of many organisms including birds and arthropods (Recher & Holmes 1985; Recher 1991; Raegan 1992; McCune & Amsberry 1997; Brokaw & Lent 1999). In jarrah forest, birds that forage in the canopy have been identified as being detrimentally affected by contemporary logging practices (Craig 1999). Fifteen attributes of canopy cover were investigated in this research (Table 4.1)

5) ***Ground Cover***

Many vertebrate and invertebrate taxa require specific attributes of ground cover (e.g., understorey vegetation, leaf litter, CWD, fine woody debris, ash beds) to feed, nest, roost and shelter on or near the ground (Facelli & Pickett 1991; Xiong & Nilsson 1999; Hooper *et al.* 2000; Zimmer 2002). The cover provided by different ground attributes have been shown to be significantly influenced by logging (Ganzhorn *et al.* 1990; Thiollay 1992; Palik & Engstrom 1999; Aleixo

1999; Fredericksen & Mostacedo 2000). Although no study has quantified how ground attributes are affected by logging in jarrah forest, many attributes have been assessed in relation to fire (e.g., Hingston 1985; Bell *et al.* 1989; Christensen & Abbott 1989; Van Huerck *et al.* 1998). For example, the cover provided by vegetation at different heights has been found to influence the composition of birds after fire in jarrah forest (Christensen & Kimber 1975). Eleven attributes of ground cover were assessed at Mt Dale (Table 4.1).

4.2.2 Sampling Program

All attributes were measured at each the 20 sites at Mt Dale (see Section 3.2). Measurements were undertaken between July 2000 and December 2001.

4.2.2.1 Composition and Decay of Trees

The stand basal area (m^2 ha^{-1}) of fifteen different attributes was calculated using the plotless method. This is a standard technique for measuring the stand basal area (m^2 ha^{-1}) of trees within a stand (Reid & Stephen 2002). It involves conducting a 360^o arc survey using an angle gauge from a stationary position. For all trees selected by the angle gauge within the arc, the diameter over bark at 1.3m above ground (Diameter at Breast Height or 'DBH') was measured using a diameter tape. Each jarrah and marri (*Corymbia calophylla*) was categorised as one of three strata levels; 'dominant' (trees that were taller, and emerged beyond, the surrounding tree canopy), 'codominant' (mature trees that had attained a height similar to the surrounding tree cover, which collectively provided the majority of canopy cover) or 'subdominant' (trees that were shorter than codominant trees) (classification as of Abbott & Loneragan 1986). Sheoak (*Allocasaurina fraseriana*) was not placed in these categories due to the

difficulties of determining whether an individual was codominant or subdominant.

At each site, four basal area sweeps were conducted (Fig. 4.2). This was achieved by dividing the site into quarters, and conducting one basal area sweep in each quarter (Fig. 4.2). The Stand Basal Area Over Bark (Stand B.A.O.B) (m^2 ha^{-1}) was calculated for all species in all strata using Equation 4.1.

EQ. 4.1

$$\text{Stand B.A.O.B } (m^2\, ha^{-1}) = \frac{\text{Tally of Stems x Basal Area Factor}}{\text{Number of sweeps}}$$

Where the Basal Area Factor = 1, and the number of sweeps = 4.

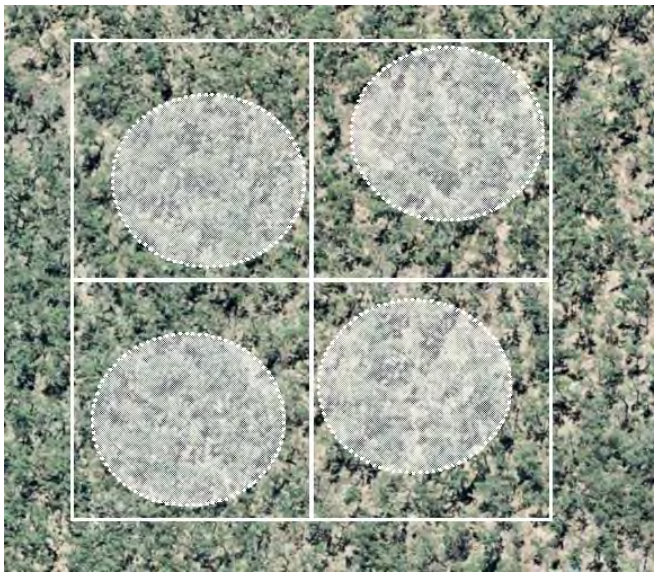


Fig. 4.2 Aerial photo of jarrah forest with 1 ha plot. A basal area sweep was conducted from a randomly selected point within each quarter.

All sampled jarrah trees were also categorised using a modified version of Whitford's Decay Classes (see Whitford & Williams 2001; Whitford 2002). This system was modified because initial surveys showed it was difficult to place

individual trees into one of the eight classes defined by Whitford (2002). This is because individual trees often had characteristics of two decay classes. As such, Whitfords Decay Classes were modified to account for this natural variability by reducing the original eight classes into four broader classes (Table 4.1; Fig. 4.3). Trees were therefore placed in one of four categories depending on branch senescence, rather than the eight specific categories used by Whitford & Williams (2001) and Whitford (2002) (Table 4.2).

Table 4.2 The relationship between the classes of decay used to classify jarrah trees in this research and the classes of decay described by Whitford & Williams (2001).

Tree Decay classes used in this research	**Tree Decay Classes defined by Whitford and Williams (2001)**
Decay Class 1	Category 1 and 2
Decay Class 2	Category 3 and 4
Decay Class 3	Category 5 and 6
Decay Class 4	Category 7 and 8

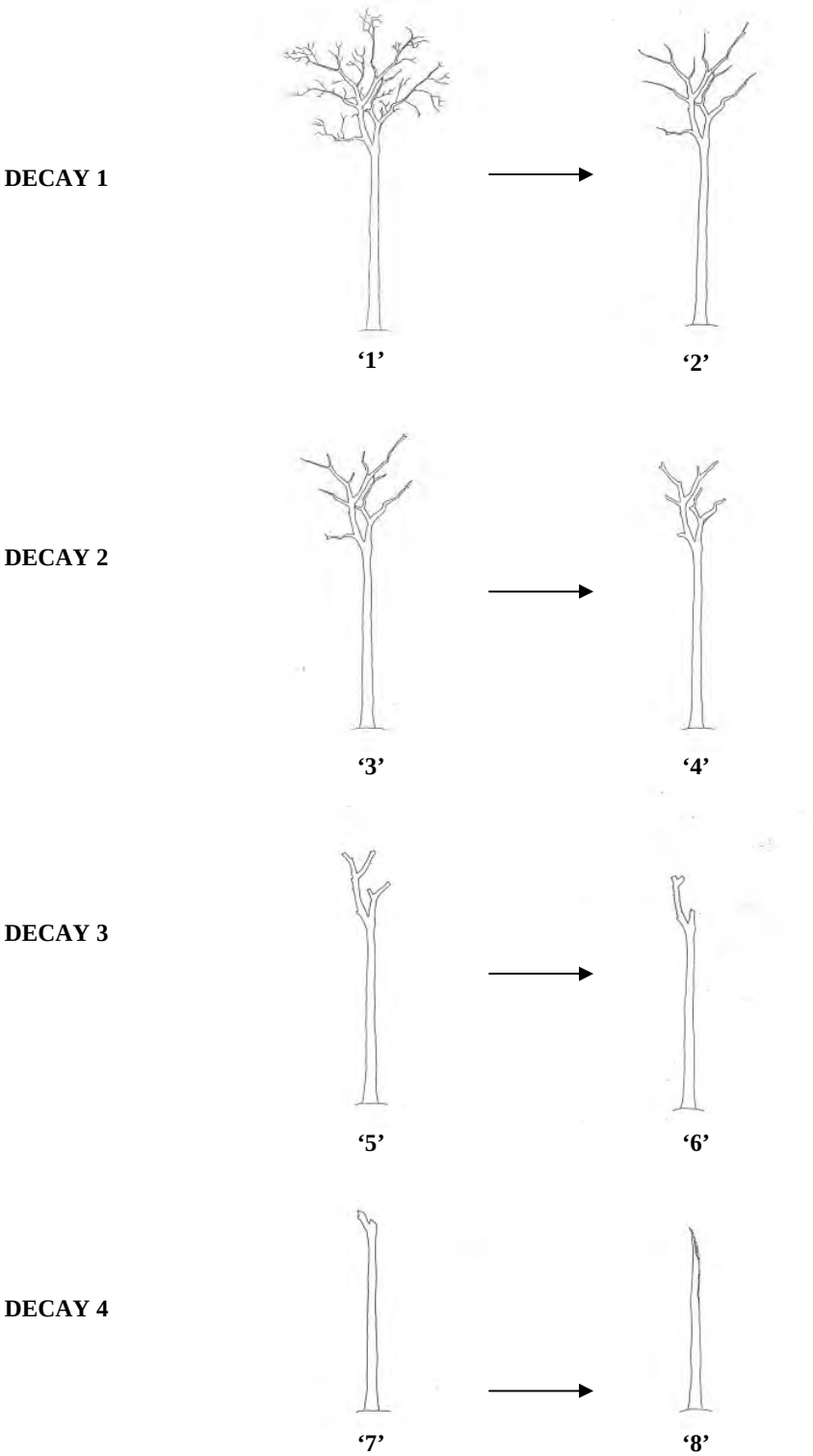


Fig. 4.3 The stages of branch decay used to categorise jarrah. 'Decay 1', 'Decay 2', 'Decay 3', and 'Decay 4', refers to the decay classes used in this research. Numbers under each tree represents Whitfords Decay Classes (Whitford & Williams 2001; Whitford 2002).

4.2.2.2 Coarse Woody Debris

Coarse woody debris (CWD) was classified into three components: (i) logs (branches and boles >10 cm diameter at one end and greater than 1 m in length), (ii) standing dead trees (stags), and (iii) stumps (Harmon & Sexton 1996).

(i) Logs

The line transect is an effective way of measuring the volume of logs in forests (Harmon & Sexton 1996). For each site, three 100 m transects were traversed (Fig. 4.4). Each transect was randomly positioned within each site using a random number table. The random number table calculated the distance between a randomly selected corner and the start of the transect. All logs that intersected the transect line were sampled and the species it originated from recorded (jarrah, marri, sheoak, banksia ***Banksia grandis*** and balga ***Xanthorrhoea preissii*** Endl.). The length and the diameter (top, middle and bottom) was recorded for each log. All jarrah logs were also placed in one of five decay classes (modified from Whitford & Williams 2001) (Table 4.3; Table 4.4). As with branch decay, pilot studies showed that placing logs into one of 10 decay classes was often difficult because the log would have characteristics of two classes. These decay classes were modified in the same was as for branch decay, that is, the ten decay classes of Whitford and Williams (2001) were reduced into five broad classes (Table 4.4).

Fig. 4.4 Aerial photo of jarrah forest with 1-ha plot. Three 100m transects were used to assess the volume of logs at each site.

Table 4.3 The characteristics used to differentiate jarrah logs into different stages of decay (modified from Whitford & Williams 2001 p. 186).

Log Characteristics	LOG DECAY SCORE 1	2	3	4	5
Bark	Intact-trace	Absent	Absent	Absent	Absent
Sapwood	Hard, grey to light brown	Grey, can be scuffed by boot, but ends hard to break off	Crumbly	Sapwood partly missing, or gone	Sapwood gone
Heartwood Strength	Hard	Hard	Decay present, can be broken away by hand	Easily broken away	Rotten
Shape of unburnt cross-section	Round	Round	75%- 95% round	25-75% round	< 25% round
Log contact with ground	Log free of ground	Log in contact with ground	Log in contact with ground for full length	Log sitting in ground	Merging with ground, with only trace evidence of log remaining
Branches	Minor branches intact	Major branch stubs	Major branch stubs may be present	None	None
Fissures	Possible cracks	Cracks present, slight fissuring, 1-2 cm deep, 8-10 cm apart	2 cm deep about 10 cm apart	3-7cm deep furrows in log surface	Deep furrows in log surface
Presence of Moss	Absent	Usually absent	Present	May be absent	May be absent

Table 4.4 The relationship between the classes of decay used to classify jarrah logs in this research and the classes of decay described by Whitford & Williams (2001).

Log Decay Classes	Categories used by Whitford and Williams (2001)
Decay Class 1	Category 1 and 2
Decay Class 2	Category 3 and 4
Decay Class 3	Category 5 and 6
Decay Class 4	Category 7 and 8
Decay Class 5	Category 9 and 10

The volume per hectare of each type of log was assessed using Equation 4.2 (from Harmon & Sexton 1996).

Eq. 4.2 $$\text{Volume } (m^3\,ha^{-1}) = \frac{9.869 \times \Sigma d^2}{8\,L}$$

Where 'd' is diameter of CWD (m) and 'L'= total transect length (= 300m)

(ii) Stags

All Decay 4 trees were considered a standing dead tree (stag) (see Table 3). The volume for each stag was calculated using Equation 4.3.

Eq. 4.3 $$\text{Volume } (m^3) = \text{Basal Area} \times \text{Height} \times \text{form factor.}$$

Where 'Basal Area' is calculated from diameter measures (m), and 'Height' estimated to the nearest metre. The form factor for jarrah was 0.6 (*sensu* Abbott & Loneragan 1986). Suming the volume of each stag provided the total volume per hectare ($m^3\,ha^{-1}$). It is important to note that this measure is an underestimate of the total stag volume for two reasons. Firstly, no marri stags were measured. Secondly, more jarrah stags would have occurred in each site than those that occurred in the area covered by the four basal area sweeps (see Fig. 4.2).

(iii) Stumps

All dead stumps were measured at each site. Stumps with coppice were not considered as they are still living elements within the ecosystem. The total volume of dead stumps was estimated by totaling the volumes calculated for each stump using equation 4.4.

Eq. 4.4 $$\text{Volume } (m^3) = BA \times H\ .$$

where BA is the basal area (m^2) of the bole removed (calculated from the circumference), and H is the height (m) of the stump. The volume of stumps was added to give a total volume per hectare.

4.2.2.3 Size Distribution of Trees and Logs

The basal area of small (0.2 - 0.6m d.o.b.), medium (0.6 - 1.0m d.o.b) and large (> 1m d.o.b) trees was determined using the basal area gauge (see Fig. 4.2). These are arbitrary size delineations; for instance, Abbott & Loneragan (1986) described 'large trees' as > 1m d.o.b. The basal area for these size classes was calculated for jarrah and 'non-jarrah' species. This is because there were insufficient trees (often < 5 trees per site) of marri and sheoak to be meaningfully divided into categories.

Logs were classed as small (between 0-0.1 m^3 in size), medium (0.1 - 1 m^3), or large (> 1 m^3). The volume of each log was calculated using Newton's Formula (Eq. 4.5), which has been shown to be the most reliable of the calculations of log volumes (Harmon & Sexton 1996).

Eq. 4.5 $$V\ (m^3) = L\ (A_b + 4\ A_m + A_t) / 6$$

Where L= the length of a log (m), and A_b, A_m and A_t are the basal area of the base, middle and top respectively (m). For each size class, the volume for the stand was calculated using Eq. 4.2

4.2.2.4 Canopy Cover

Canopy cover was estimated from 200 randomly selected points per site using a crownometer. These points were located by starting from a random location, and

moving 5 m in a random direction to the next point. The random location was located by placing two 100m tapes on two perpendicular sides of the site. Two random numbers were then selected to give a value in metres for each tape (i.e., a '5' and '3' = 53m). The two 100m tapes were then used as a grid system, with the two randomly derived numbers becoming the co-ordinates for the first point where canopy cover was measured. The second point was then located randomly using a stick with a taped end. This stick was spun in the air, and where it landed provided a random direction. The second point was 10m away from the first point in the randomly provided direction. This technique of spinning the stick and walking ten metres was repeated until 200 sample points were attained. If a small area was repeatedly surveyed, or the random direction placed me beyond the sites boundaries, a new starting position was determined, and the process started again.

At each point, where the cross-hairs of a crownometer hit a component of a tree (e.g., leaf or branch), the species and strata level were recorded. For the dominant strata, a total cover was calculated as well as individual cover for jarrah and marri. For the codominant strata, sheoak was added to these species. Subdominant species consisted of all aforementioned species, as well as 'Banksia' (*B. grandis*) and 'Other' (*Persoonia longifolia, P. elliptica, Dryandra sessilis* and *Nuytsia floribunda*).

The structural diversity of the canopy was calculated using the Shannon-Wiener Index. For each site, the total dominant, codominant and subdominant were summed, and the proportion (p) of each strata calculated (Murdoch *et al.* 1972). An Index (H) for each site was calculated using equation 4.6.

Eq. 4.6 $H' = - \Sigma (p \ln p)$

4.2.2.5 Ground Cover

Ground cover was assessed using a Levee Pole at each of the 200 points described in 4.2.2.4. Ground cover comprised the surface interface characteristics where the Levee Pole encountered the ground. There were six categories of ground cover:

1. **Leaf Litter/ Fine Woody Debris**: cover that consisted of leaf litter, twigs, or any other organic material that comprises fine woody debris.
2. **Exposed Laterite:** cover consisting of laterite cap rock.
3. **Woody Debris:** cover of branches and 'logs' (defined as >10cm diameter).
4. **Bare Soil:** no cover above the soil.
5. **Stump Cover:** cover provided by dead stumps.
6. **Other Ground Cover:** cover that included ashbeds, moss, termite mounds and stump holes.

The vertical cover provided by understorey plants was calculated using the vertical stratifications of the levee pole. This 2 m pole was split into four sections (0-20cm, 20-40 cm, 40-100 cm, 100-200 cm height from the ground) (modified from Naveh & Whittaker 1979). For each of the 200 points, a tally was kept for wherever vegetation touched these sections, and a percentage cover calculated. Foliage height diversity was measured using Shannon-Wiener Index H '(Murdoch *et al.* 1972). The cover of the four vertical sections was summed, and the proportion (p) that each strata contributed was calculated. An Index for each site was then determined using Equation 4.6.

4.2.7 Statistical Procedure

Univariate Analysis

Analysis of Variance (ANOVA) were performed on each of the 65 attributes using the SPSS statistical package (Illinois, USA). There are three major assumptions of ANOVA. The first is that observations (strictly speaking, the 'error terms') are independent (Cochran 1947). The randomisation of sites within and between treatments meets the requirements of this assumption (as outlined in Chapter 2). The second assumption is that observations (after transformation) are normally distributed (Underwood 1997). Prior to ANOVA analysis, normality was tested using SPSS. The third assumption, that compared groups have equal variances, was tested using Levene's statistic using the SPSS package at alpha = 0.01 (as recommended by McGuiness 2002). As recent evidence shows that ANOVAs are robust against moderate violations of this assumption (see McGuiness 2002), ANOVAs were conducted if compared groups had unequal variances with the precaution that the data are carefully scrutinised for outliers if significant differences were found in the ANOVA. This is because heterogeneous variances between compared groups increases the frequency of Type 1 errors, and as such, a non-significant F test is likely to be reliable whether or not variances are homogeneous (McGuinness 2002).

A cautious approach was taken with regards to interpreting each ANOVA (as recommended by Mapstone 1996). Past research in jarrah forest has had low statistical power, making it difficult to reject the null hypothesis if it is truly false and therefore increasing the chances committing a Type II error (i.e., accepting Ho when it is truly false) (Strehlow ***et al.*** 2002; Calver 2003). Committing a Type 2 error could result in the loss of biodiversity because deleterious

management practices are not recognised (Di Stefano 2001; Calver 2003). In my research, the null hypothesis (Ho) and alternative hypothesis (Ha) for each attribute ('X') were:

Ho : There is no difference in 'X' among the five treatments.

Ha : There is a difference in 'X' among the five treatments.

In this research, accepting Ho when it is false could result in logging disturbance adversely affecting biodiversity in jarrah forest. The following two steps were taken for each ANOVA to ensure that the chance of making a Type 11 error was reduced:

1) When the probability value was less than 5 % (i.e., $p < 0.05$), then alternate hypothesis was accepted.

2) When the probability value was between 5 and 10% ($0.05 < p < 0.1$), the statistical power of the ANOVA was scrutinised. If the test did not have the statistical power to reject the null hypothesis if it was truly false (power < 0.8 ***sensu*** Strehlow ***et al.*** 2002) than the probability value for the ANOVA was increased from 0.05 to 0.l. This change decreases the risk of making a Type 11 error but increases the risk of making a Type 1 error (i.e., accepting Ha when it is truly false) (Calver 2003).

Where significant differences were found in the ANOVA, ***post hoc*** analysis comparing between means was analysed using Games-Howell statistics as the variances were not assumed to be equal.

Multivariate Analysis

The five structural categories ('composition and decay of trees', 'coarse woody debris', 'size distribution of trees and logs', 'canopy cover' and 'ground cover') were analysed separately. Prior to analysis, all attributes were transformed (square root, fourth root, log and Arcsine for percentages) to remove any skew in the data (as recommended by Clarke & Warwick 1994). The data were then correlated in a Draftman's Analysis to assess for colinearity between variables using the PRIMER 5, v 2.1 (Plymouth, UK). One member of each of the collinear variables (> 0.6) was removed from further analysis. Where attributes were collinear because one attribute contained a 'total' of another, the 'total' variable was removed (e.g., 'total emergent stand BA' was removed because it consists of 'jarrah stand BA' and 'marri stand BA').

Multivariate techniques were used to assess whether structural attributes differed among the five treatments. One –way ANOSIMs, using the Standardised Euclidean Matrix, assessed whether there were any differences in the 'composition' of structural attributes between treatments (as recommended by Clarke & Gorley 2001). For treatments that were found to be statistically significant in the ANOSIM, a SIMPER analysis was performed to assess which attributes contributed most to the differences. Attributes that contributed more than 5% were then tabulated to show how the variation was partitioned among treatments. Ordination using non-metric multidimensional scaling (nMDS) plots were used to graphically depict the similarity between sites.

4.3 RESULTS

4.3.1 Stand Basal Area

The stand basal area (SBA) of all trees > 20 cm d.o.b. ranged between 10.5 - 26.8 m^2 ha^{-1}. The majority of trees were codominant (39 - 84 % of the SBA measured at each site), with a smaller proportion (6 - 24 %) emerging beyond this strata (dominant trees) or being subdominant to it (13 - 52 %). Over 50 % of SBA were jarrah of decay class 1 (17 - 62 %) or decay class 2 (17 – 50 %). A smaller proportion of jarrah was decay class 3 (2 - 18 %) or decay class 4 (0 - 6 %). The SBA of marri (1 - 26 %) and sheoak (0 – 33 %) was a variable component of the total SBA found at each site.

Univariate Analysis

Prior to Analysis of Variance calculations, three variables (the SBA of marri dominant, jarrah decay 3 and sheoak) were transformed using natural log. Five of the 15 attributes assessed had significantly different SBAs among the treatments ($p < 0.05$; Table 4.5). These attributes were total SBA, total dominant SBA, jarrah dominant SBA, jarrah decay 2 SBA and jarrah decay 4 SBA (Table 4.5). At standard levels of significance, there were no differences in SBA for the other 10 attributes ($p > 0.05$; Table 4.5). When alpha was adjusted to account for low power (i.e., probability was changed from 0.05 to 0.1 when statistical power was less than 0.8), there was also no difference in the SBA for these 10 attributes among different treatments ($p > 0.1$; Table 4.5).

Table 4.5 Summary of one-way ANOVAs for 15 attributes of stand basal area (df = 19). Statistical power is also presented. Highlighted rows indicate those attributes that had significantly different stand basal area between five treatments representing different disturbance regimes. '*' represents attributes that were transformed by natural log prior to conducting the ANOVA. Dashed lines differentiate subsets of attributes that are plotted on the same figure (see Fig. 4.5; Fig. 4.6 and Fig. 4.7).

Attributes	*F*	P value	Observed Power
Total	3.23	.04	0.69
Total Dominant	3.21	.04	0.68
Total Codominant	1.23	.34	0.29
Total Subdominant	1.38	.29	0.33
Jarrah Dominant	3.06	.05	0.66
Marri Dominant *	0.67	.62	0.17
Jarrah Codominant	0.88	.50	0.22
Marri Codominant	2.28	.11	0.52
Jarrah Subdominant	1.32	.31	0.31
Marri Subdominant	0.60	.67	0.16
Total Sheoak *	0.81	.53	0.21
Jarrah Decay 1	1.23	.33	0.30
Jarrah Decay 2	5.74	.01	0.92
Jarrah Decay 3 *	1.90	.16	0.44
Jarrah Decay 4	3.07	.04	0.67

Pairwise analysis of total SBA showed that both gap logged treatments ('gap logged 1990' and 'gap logged/burnt 2000') had lower SBA than the other three treatments ($p < 0.05$; Fig. 4.5). Total SBA was similar in unlogged, burnt 2000 and selectively logged prior to 1950 treatments ($p > 0.05$; Fig. 4.5). With respect to different strata levels, the SBA of dominant trees was lower in treatments disturbed by gap logging than the two unlogged treatments ('undisturbed' and 'burnt 2000') ($p < 0.05$; Fig. 4.5). There were no differences between the SBA of dominant trees in forest selectively logged prior to 1950 and the other four treatments ($p > 0.05$; Fig. 4.5). There were also no differences in the SBA of codominant and subdominant trees among the five treatments ($p > 0.1$; Table 4.5; Fig. 4.5).

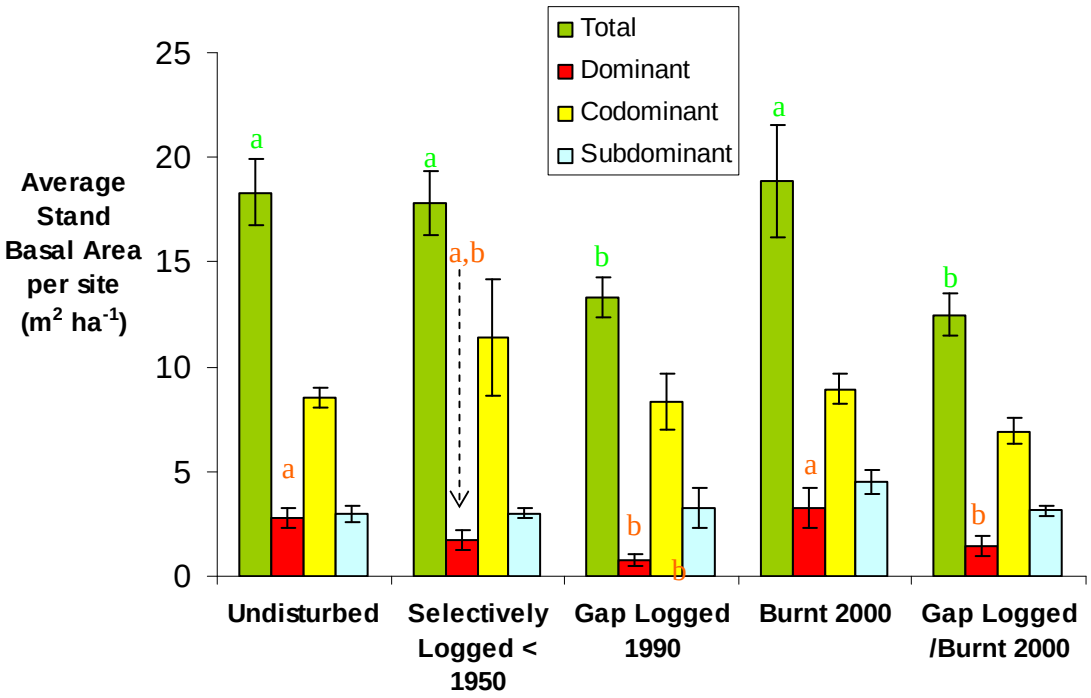


Fig. 4.5 The average (± SE) stand basal area of all trees, dominant trees, codominant trees, and subdominant trees per site for five treatments representing different disturbance regimes. Different letters indicate significant differences (see Table 4.5).

The low SBA of dominant trees in forest disturbed by gap logging was attributed to lower SBA of dominant jarrah trees ($p < 0.05$). This is because the SBA of marri dominant trees was not significantly different among the five treatments ($p > 0.1$; Table 4.5; Fig. 4.6). The SBA of codominant jarrah, codominant marri, subdominant jarrah, subdominant marri and sheoak was also not different among the five treatments ($p > 0.1$; Table 4.5; Fig. 4.6).

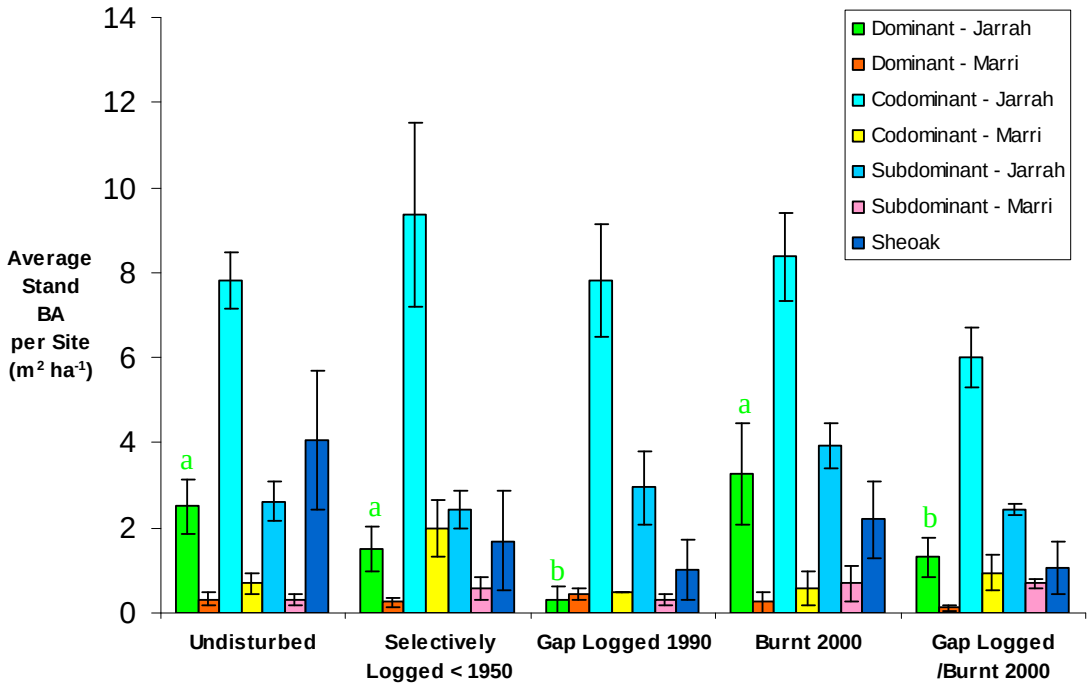


Fig. 4.6 The average (± SE) stand basal area of dominant, codominant and subdominant jarrah and marri per site for five treatments representing different disturbance regimes. The average stand basal area of sheoak, which was not classified into different strata categories, is also presented. Different letters indicate significant differences (see Table 4.5).

With respect to jarrah tree decay, the SBA of two decay classes was different among the five treatments ($p < 0.05$; Table 4.5). The SBA of decay 2 trees was lower in both gap logged treatments compared to both unlogged treatments ($p < 0.05$), but was similar between gap logged treatments and forest selectively logged prior to 1950 ($p > 0.05$; Fig. 4.7). Forest gap logged in 1990 also had lower SBA of decay 4 trees than all other treatments ($p < 0.05$) (Fig. 4.7). The SBA of decay 1 and decay 3 trees were not statistically significant among treatments ($p > 0.1$; Fig. 4.7).

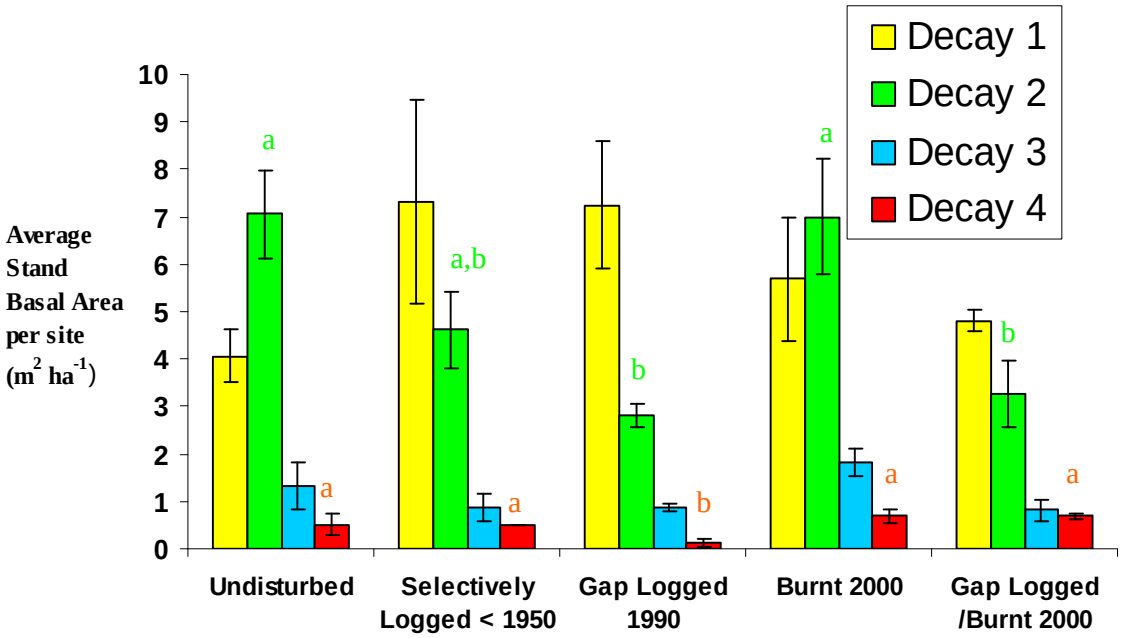


Fig. 4.7 The average (± SE) stand basal area of jarrah trees per site for five treatments representing different disturbance regimes. Jarrah trees were categorised into four decay classes (see Fig. 4.3). Different letters indicate significant differences (see Table 4.5).

Multivariate Analysis

Prior to multivariate analysis, three attributes (jarrah codominant, total codominant and total subdominant) were removed because they were collinear with other attributes (correlation coefficient > 0.8). Multivariate analysis of the remaining 11 attributes revealed there were differences in attributes of SBA attributes among treatments (Global R = 0.208, $p < 0.01$). The two unlogged treatments (‘undisturbed’ and ‘burnt 2000 'forest) were significantly different to the two gap logged treatments (gap logged in 1990 and gap logged/burnt 2000) (Table 4.6). Forest gap logged in 1990 had distinct SBA attributes than the other four treatments on the ordination (Fig. 4.8). Sites that were gap logged and burnt in 2000 also had distinct attributes, but were more closely related to forest burnt in 2000 and forest selectively logged prior to 1950 than sites that were gap logged in 1990 (Fig. 4.8). Sites that had never been logged also appeared to have a

greater variability in attributes of SBA than forest that had been disturbed by gap logging (Fig. 4.8). Similarly, sites in forest selectively logged prior to 1950 were more variable than forest that had been disturbed by gap logging (Fig. 4.8).

Table 4.6 Summary of ANOSIM pairwise analysis testing whether the composition of attributes that comprised stand basal area were different between five treatments representing different disturbance regimes. Highlighted rows represent pairs of treatments that were significantly different ($p < 0.05$).

Groups	Global *R*	P
Undisturbed, Burnt 2000	-0.094	0.714
Undisturbed, Gap Logged/ Burnt 2000	0.313	0.029
Undisturbed, Gap Logged 1990	0.427	0.029
Undisturbed, Selectively Logged <1950	0.01	0.457
Burnt 2000, Gap Logged/ Burnt 2000	0.323	0.029
Burnt 2000, Gap Logged 1990	0.531	0.029
Burnt 2000, Selectively Logged <1950	0.083	0.343
Gap Logged/ Burnt 2000, Gap Logged 1990	0.181	0.057
Gap Logged/ Burnt 2000, Selectively Logged <1950	0.073	0.343
Gap Logged 1990, Selectively Logged <1950	0.292	0.114

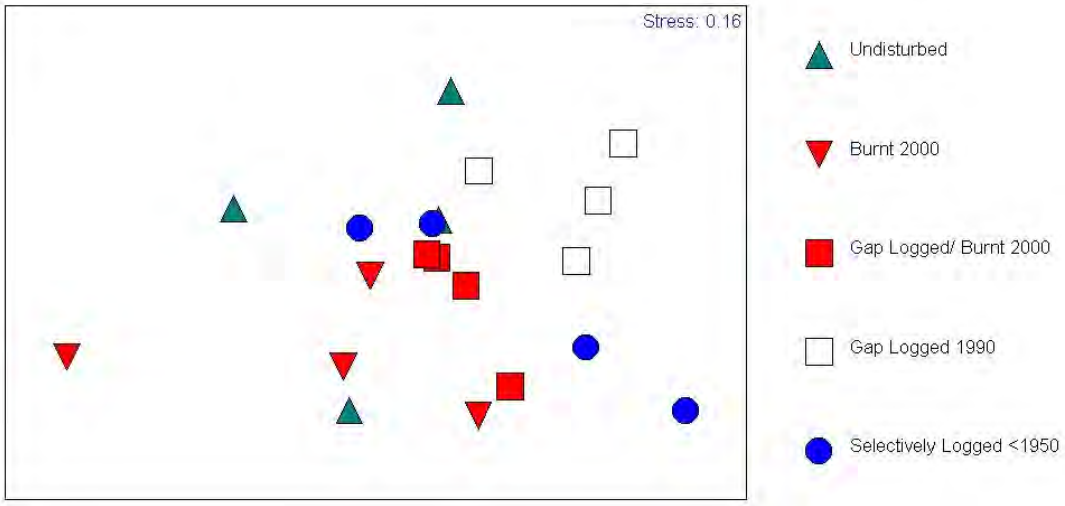


Fig. 4.8 Non-metric multidimensional scaling plot showing the effect of five disturbance regimes on attributes that comprise forest stand basal area.

SIMPER analysis found that the greater SBA of all trees and the SBA of jarrah decay 2 trees in unlogged treatments ('undisturbed' and 'burnt 2000') contributed

the most to differences between gap logged treatments ('gap logged 1990' and 'gap logged/ burnt 2000') and unlogged treatments (Table 4.7). The SBA of jarrah dominant contributed less significantly to differences between gap logged treatments and unlogged treatments (Table 4.7). In addition these three attributes, the greater SBA of sheoak trees in undisturbed forest also contributed to difference in undisturbed and forest gap-logged/ burnt in 2000 (Table 4.7).

Table 4.7 Attributes of forest stand basal area that contributed to differences found between treatments identified as significant by pairwise ANOSIM analysis (see Table 4.6). Attributes listed are those that contributed greater than 5% of the difference found between treatments as determined by SIMPER analysis. Attributes are grouped within the 'treatment' in which they had greater BA (m^2 ha^{-1}), and ranked according to their contribution to difference between treatments.

Treatments	**Attributes contributing to difference (% of variation contributed by attribute)**
Undisturbed	Total (36.8) Jarrah Decay 2 (24.3) Jarrah Dominant (9.8) Sheoak (6.9)
Gap Logged/ Burnt 2000	-
Undisturbed	Total (26.6) Jarrah Decay 2 (22.4) Jarrah Dominant (10.6)
Gap Logged 1990	Jarrah Decay 1 (18.7)
Burnt 2000	Total (34.7) Jarrah Decay 2 (21.1) Jarrah Dominant (10.8)
Gap Logged/ Burnt 2000	-
Burnt 2000	Total (27.2) Jarrah Decay 2 (21.1) Jarrah Dominant (15.4)
Gap Logged 1990	Jarrah Decay 1 (13.4)

4.4.2 Coarse Woody Debris

Logs accounted for the majority of CWD at each site (between 62 - 97% of the total volume of CWD). Standing dead trees comprised a much smaller portion of CWD (0 - 36%), and dead stumps were associated with logged sites (0 – 10 %). The majority of logs were jarrah (79 – 98% of the total volume of logs per site). Of the total volume of jarrah logs per site, the contribution of each decay class varied significantly between sites (decay class 1: 1 – 70 % of the volume of jarrah logs per site; decay class 2: 2 – 57 %; decay class 3: 2 - 61 % ; decay class 4: 1 – 78 % ; decay class 5: 1 – 23 %).

Univariate Analysis

Prior to one way Analysis of variance, eight (the volume of jarrah decay '1', decay '4' and decay '5' logs, banksia logs, sheoak logs, marri logs, balga logs and stags) of the 15 CWD attributes assessed were transformed using natural log. Five of these attributes had significantly different volumes (m^3 ha $^{-1}$) of CWD among the treatments at standard levels of significance ($p < 0.05$; Table 4.8). These were the total CWD, all logs, stumps, all jarrah logs and all decay 1 jarrah logs (Table 4.8). When alpha was adjusted to account for low power, the volume of balga logs was different among treatments ($p < 0.1$; Table 4.8).

Table 4.8 Summary of one way ANOVAs for 15 attributes of coarse woody debris (df = 19). Statistical power is also presented. Highlighted rows indicate those attributes that had significantly different volume of coarse woody debris between treatments representing different disturbance regimes (dark grey $p < 0.05$; light grey $p < 0.1$). '*' represents attributes that were transformed by natural log prior to conducting the ANOVA. Dashed lines differentiates subsets of attributes that are plotted on the same figure (see Fig. 4.9; Fig. 4.10 and Fig. 4.11).

Attributes	*F*	P value	Observed Power
Total	7.60	.00	0.98
Logs	10.12	.00	0.99
Stumps	7.25	.00	0.97
Stags *	1.58	.23	0.37
Jarrah Logs	9.23	.00	0.99
Decay 1 *	16.02	.00	1.00
Decay 2	0.63	.65	0.16
Decay 3	2.22	.12	0.51
Decay 4 *	2.37	.11	0.61
Decay 5 *	1.03	.43	0.25
All non-jarrah Logs	0.36	.83	0.11
Banksia Logs *	0.61	.66	0.16
Sheoak Logs *	0.15	.96	0.07
Marri Logs *	0.20	.93	0.08
Balga Logs *	2.81	.06	0.67

Pairwise analysis showed the total volume of CWD was greater in both gap logged treatments than in the other three treatments ($p < 0.05$; Fig. 4.9). There were no differences in the total volume of CWD among the three treatments that were not disturbed by gap logging ($p > 0.05$; Fig. 4.9). Of the three broad categories that together comprise CWD (logs, stumps and stags), the volume of logs and stumps were different among the five treatments ($p < 0.05$; Table 4.8). Pairwise analysis showed the volume of logs was greater in forest disturbed by gap logging than in the other three treatments ($p < 0.05$; Fig. 4.9). The volume of stumps was greater in the three treatments that had been disturbed by logging ('gap logged 1990', 'gap logged/ burnt 2000' and 'forest selectively logged < 1950') than in the two treatments that had not been logged ('undisturbed' and 'burnt 2000') ($p < 0.05$; Fig. 4.9). There were no statistical differences in the volume of stags among treatments ($p > 0.1$; Table 4.8; Fig. 4.9).

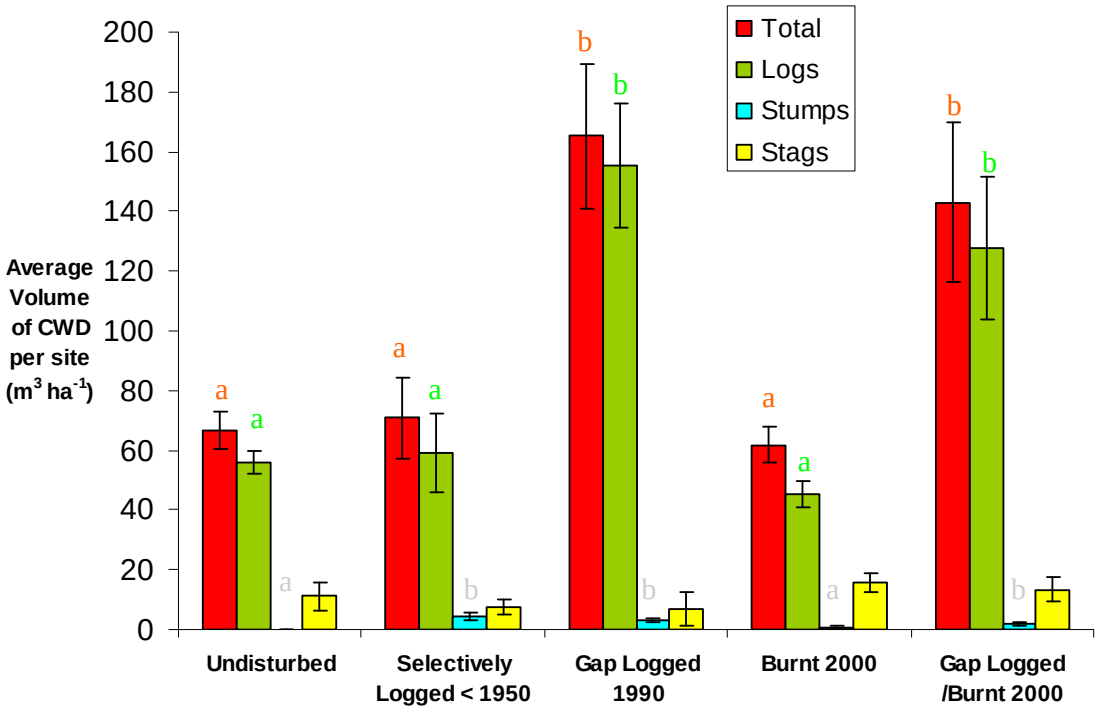


Fig. 4.9 The average (± SE) volume of the total coarse woody debris, logs, stags and stumps per site for five treatments representing different disturbance regimes. Different letters indicate significant differences (see Table 4.8)

With respect to the log component of CWD, there was a greater volume of jarrah logs in the two treatments disturbed by gap logging ('gap logged 1990' and 'gap logged/burnt 2000') than the other three treatments ($p < 0.05$; Table 4.8; Fig. 4.10). Of the five decay classes used to differentiate jarrah logs, the volume of decay 1 jarrah logs was the only decay class found to be significantly different among the five treatments ($p < 0.05$; Table 4.8). There was a greater volume of jarrah logs of decay class 1 in forest disturbed by gap logging compared to the other three treatments ($p < 0.05$; Fig. 4.10).

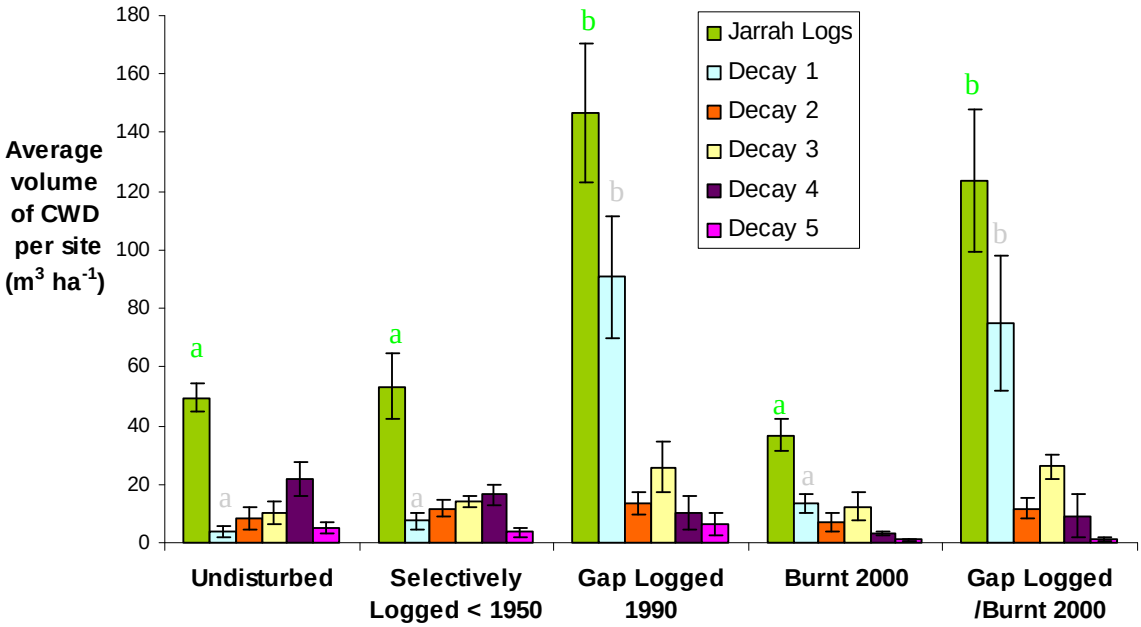


Fig. 4.10 The average (± SE) volume jarrah logs per site for five treatments representing different disturbance regimes for each treatment. Jarrah logs were categorised into five decay classes. Different letters indicate significant differences (see Table 4.8).

With respect to non-jarrah logs, there were no differences in the volume of non-jarrah logs among the five treatments at standard levels of significance ($p > 0.05$; Table 4.8). With regards to the volume of banksia logs, sheoak logs and marri logs, there was large variation in volumes between sites within each treatment (Fig. 4.11). When alpha was adjusted to account for low power, there were significant differences in the volume of balga logs among treatments ($p < 0.1$; Table 4.8). Pairwise analysis show there was a lower volume of balga logs in undisturbed forest and forest selectively logged prior to 1950 than in the other three treatments ($p < 0.1$; Fig. 4.11).

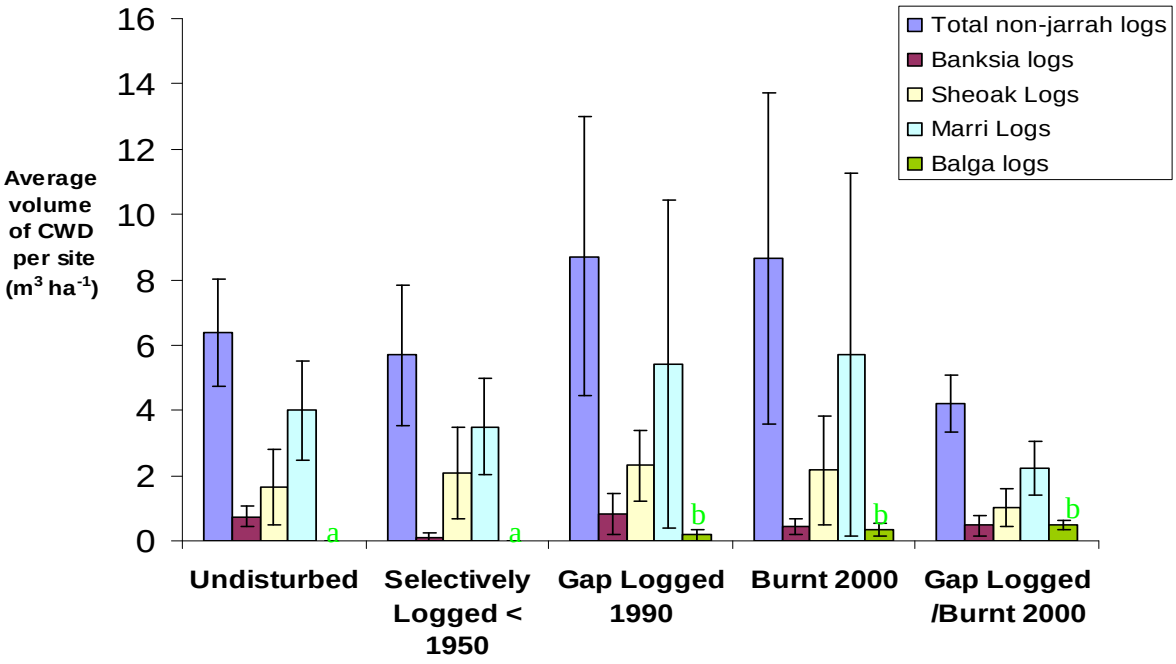


Fig. 4.11 The average (± SE) volume of non -jarrah logs per site for five treatments representing different disturbance regimes each treatment. Non - jarrah logs consisted of marri, balga, sheoak or banksias logs. Different letters indicate significant differences (see Table 4.8).

Multivariate Analysis

Total CWD, total volume of logs and total jarrah logs were collinear with other attributes and removed from multivariate analysis (correlation coefficient > 0.8). Analysis of Similarity showed there were different attributes that comprise CWD among all treatments (Global R = 0.172; $p < 0.01$). Pairwise analysis revealed that all logged forest ('selective logged < 1950', 'gap logged 1990' and 'gap logged/burnt 2000' treatments) had significantly different attributes of CWD than unlogged treatments ('undisturbed' and 'burnt 2000') (Table 4.9). There were no differences in CWD attributes between undisturbed forest and forest burnt in 2000 (Table 4.9). Similarly, there were no differences in CWD attributes between the two gap logged treatments (Table 4.9).

Table 4.9 Summary of ANOSIM pairwise analysis testing whether the composition of attributes that comprised coarse woody debris were different between five treatments representing different disturbance regimes. Highlighted rows represent pairs of treatments that were significantly different ($p < 0.05$).

Groups	Global *R*	P value
Undisturbed, Burnt 2000	0.156	0.229
Undisturbed, Gap Logged/ Burnt 2000	0.615	0.029
Undisturbed, Gap Logged 1990	0.552	0.029
Undisturbed, Selectively Logged <1950	0.594	0.029
Burnt 2000, Gap Logged/ Burnt 2000	0.323	0.029
Burnt 2000, Gap Logged 1990	0.531	0.029
Burnt 2000, Selectively Logged <1950	0.521	0.029
Gap Logged/ Burnt 2000, Gap Logged 1990	0.063	0.314
Gap Logged/ Burnt 2000, Selectively Logged <1950	0.5	0.029
Gap Logged 1990, Selectively Logged <1950	0.396	0.029

Forest disturbed by gap logging (‘gap logged 1990’ and ‘gap logged/burnt 2000’) contained different attributes of CWD than forest not disturbed by gap logging (‘undisturbed’, ‘burnt 2000’ and ‘selectively logged < 1950’) on the ordination (Fig. 4.12). Furthermore, attributes in forest selectively logged were more similar attributes to unlogged treatments (‘undisturbed’ and ‘burnt 2000’) than forest had been disturbed by gap logging (Fig. 4.12). In addition, forest gap logged in 1990, and to a lesser degree forest gap logged and burnt in 2000, had a more variable composition of CWD attributes than the other three treatments on the ordination (Fig. 4.12). In contrast, sites that were located in undisturbed forest, forest burnt in 2000 or forest selectively logged prior to 1950 appeared to contain relatively similar attributes of CWD.

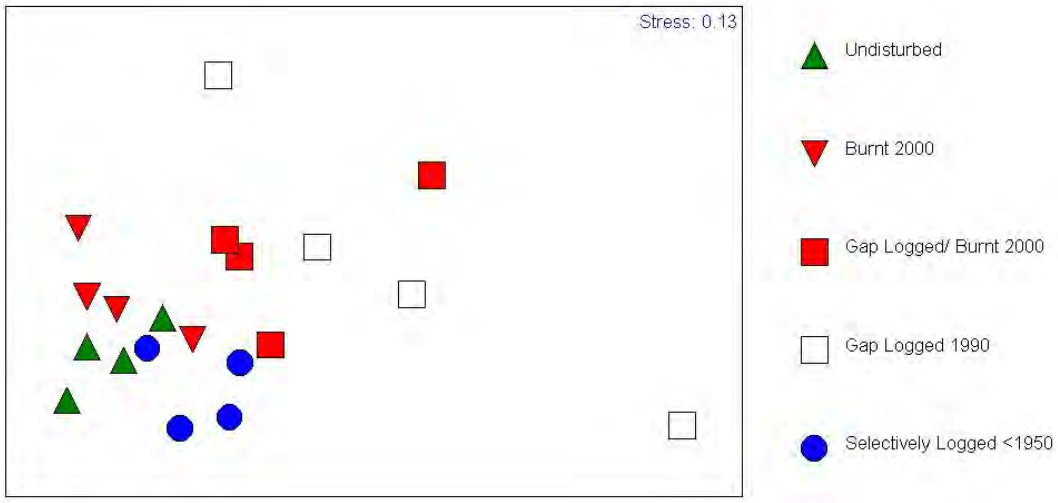


Fig 4.12 Non-metric multidimensional scaling plot showing the effect of five disturbance regimes on attributes that coarse woody debris.

The attribute that contributed most to differences between forest disturbed by gap logging ('gap logged/ burnt 2000' and 'gap logged 1990' treatments) and forest not disturbed by gap logging ('undisturbed', 'burnt 2000' and 'selectively logged < 1950 treatments') was the volume of jarrah logs of decay class 1 (Table 4.10). Greater volumes of jarrah logs of decay class 3, and to a lesser degree decay class 2, in forest disturbed by gap logging also contributed to differences among these treatments (Table 4.10). In contrast, the greater volume of decay class 4 jarrah logs in undisturbed forest and selectively logged forest contributed marginally to differences between these treatments and forest disturbed by gap logging (Table 4.10).

Table 4.10 Attributes of coarse woody debris that contributed to differences found between treatments identified as significant by pairwise ANOSIM analysis (see Table 4.9). Attributes listed are those that contributed greater than 5% of the difference found between treatments as determined by SIMPER analysis. Attributes are grouped within the treatment in which they had greater volume (m^3 ha^{-1}), and ranked according to their contribution to difference between treatments.

Treatments	**Attributes contributing to difference (% of variation contributed by attribute)**
Undisturbed	Decay 4 (12.6)
Gap Logged /Burnt 2000	Decay 1 (32.7) Decay 3 (23.4) Decay 2 (6.0)
Undisturbed	Decay 4 (10.5)
Gap Logged 1990	Decay 1 (26.9) Decay 3 (10.0) Decay 2 (8.2)
Undisturbed	-
Selectively Logged < 1950	Decay 3 (25.9) Decay 2 (24.8) Dead Stumps (17.6)
Burnt 2000	-
Gap Logged / Burnt 2000	Decay 1 (35.1) Decay 3 (26.2) Decay 2 (8.3)
Burnt 2000	-
Gap Logged 1990	Decay 1 (37.7) Decay 3 (22.1) Decay 2 (18.2)
Burnt 2000	-
Selectively Logged < 1950	Decay 3 (27.9) Decay 2 (25.5) Dead Stumps (15.3)
Selectively Logged < 1950	Decay 4 (17.5) Dead Stumps (14.3)
Gap Logged/ Burnt 2000	Decay 1 (30.7) Decay 3 (8.9) Decay 2 (5.9)
Selectively Logged < 1950	Decay 4 (12.8) Dead Stumps (6.2)
Gap Logged 1990	Decay 1 (25.7) Decay 3 (22.1) Decay 2 (6.6)

The differences in CWD attributes between unlogged forest ('undisturbed' and 'burnt' treatments) and forest selectively logged < 1950 (Table 4.9) were due to greater volumes of jarrah decay 3 logs, jarrah decay 2 logs, and dead stumps, in the forest that had been selectively logged prior to 1950 (Table 4.10). Moreover, the greater volume of dead stumps in forest selectively logged prior to 1950 also contributed to differences with this treatment and forest disturbed by gap logging ('gap logged 1990' and 'gap logged/burnt 2000')(Table 4.9).

4.4.3 Size Class of Logs and Trees

Jarrah dominated all size classes for logs (small: 9 - 100 % of total volume per site, medium: 42 - 100 %, large: 70 - 100 %) and trees (small: 22 - 95 %, medium: 61 – 100 %, large: 36 – 100 %). There was significant variation among sites in relation to the SBA of different size classes of trees and volume of different size classes of logs. Generally, the majority of logs of both jarrah and non-jarrah origin were either medium (7 - 77 %) or large (0 - 80 %) rather than small (7 - 42 %) in size. In contrast, the majority of trees were small (20 - 75 %) in size rather than medium (14 - 48 %) or large (0 - 32 %).

Univariate Analysis

Eight of the twelve attributes were transformed prior to Analysis of Variance calculations (large jarrah logs; small, medium and large non-jarrah logs; large jarrah trees; and, small, medium, and large non-jarrah trees). With respect to the six log attributes, two were significantly different among treatments at standard levels of significance ($p < 0.05$; Table 4.11). The volume of medium logs was higher in the three treatments that had been logged ('gap logged 1990', 'gap logged/burnt 2000' and 'selectively logged < 1950') compared to the two unlogged treatments ('undisturbed' and 'burnt 2000')($p < 0.05$; Fig. 4.13). The

volume of large jarrah logs was higher in the two treatments that had been disturbed by gap logging than in the other three treatments that had not been disturbed by gap logging ($p < 0.05$; Fig. 4.13). The volume of small jarrah logs and all size classes for non-jarrah logs were not statistically different among treatments when alpha was adjusted to account for low power ($p > 0.1$; Table 4.11; Fig. 4.13).

Table 4.11 Summary of one-way ANOVAs for 6 size classes of logs and 6 size classes of trees (df = 19). Statistical power is also presented. Logs were categorised as small (0 - 0.1m^3), medium (0.1 – 1m^3) or large (> 1m^3). Trees were categorised as small (20 – 60cm d.o.b.), medium (60 – 100cm d.o.b.) or large (> 100cm d.o.b.). Highlighted rows indicate those attributes that had significantly different volume of coarse woody debris among treatments (dark grey $p < 0.05$; light grey $p < 0.1$). '*' represents attributes that were transformed by natural log prior to conducting the ANOVA. Dashed lines differentiates subsets of attributes that are plotted on the same figure (see Fig. 4.13; Fig. 4.14).

Attributes	***F***	**P value**	**Observed Power**
Jarrah Logs – Small	1.61	.22	0.38
Jarrah Logs – Medium	7.76	.00	0.98
Jarrah Logs – Large *	9.97	.00	0.99
Non-jarrah Logs – Small *	0.04	.99	0.05
Non-jarrah Logs – Medium *	2.41	.10	0.55
Non-jarrah Logs – Large *	1.00	.44	0.24
Jarrah Trees – Small	1.30	.31	0.31
Jarrah Trees – Medium	3.43	.04	0.72
Jarrah Trees – Large *	2.84	.06	0.63
Non-jarrah Trees – Small *	1.31	.31	0.31
Non-jarrah Trees – Medium *	1.77	.19	0.41
Non-jarrah Trees – Large *	0.99	.44	0.24

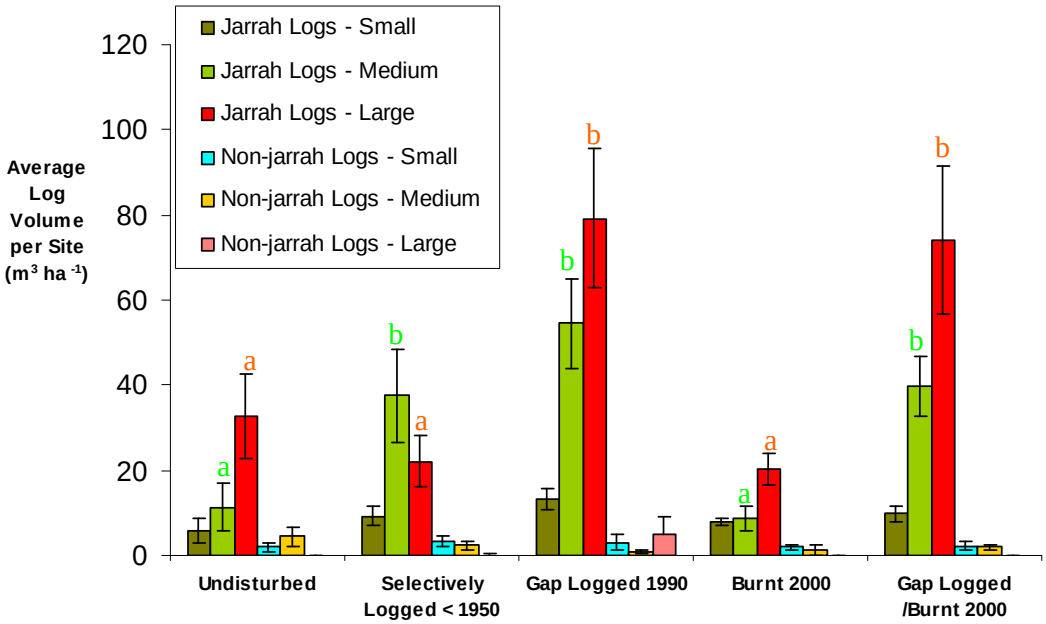


Fig. 4.13 The average (± SE) volume of jarrah and non-jarrah logs per site for five treatments representing different disturbance regimes. Logs were categorised into three different size classes; small (0 - 0.1m^3), medium (0.1 - 1m^3) or large (> 1m^3). Different letters indicate significant differences (see Table 4.11).

The only size class of tree that was significantly different among treatments at standard levels of significance was the SBA of medium jarrah trees ($p < 0.05$; Table 4.11). There was greater SBA of medium jarrah trees in the unlogged forest than in either of the two gap logged treatments ($p < 0.05$; Fig. 4.14). There was no difference in the SBA of medium jarrah trees between forest selectively logged prior to 1950 and the other four treatments ($p > 0.05$; Fig. 4.14). When alpha was adjusted to account for low statistical power for the other attributes, the SBA of large jarrah trees had similar trends to the SBA of medium jarrah trees ($p < 0.1$; Fig. 4.14).

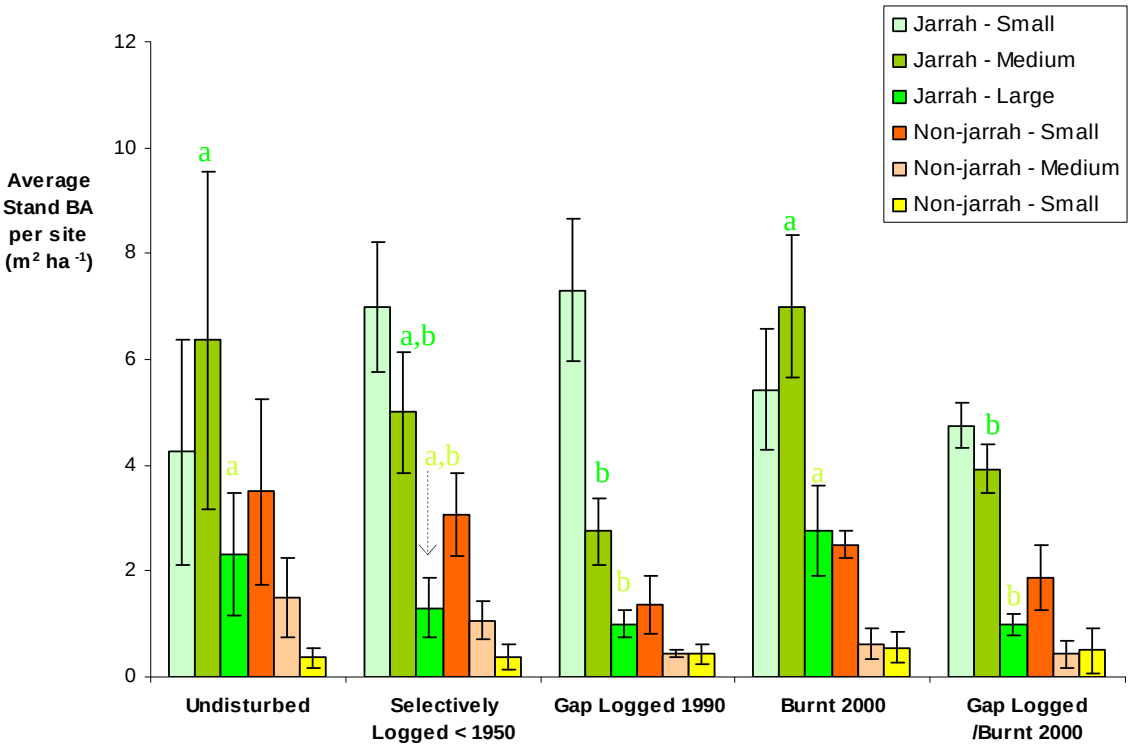


Fig. 4.14 The average (± SE) volume of jarrah and non-jarrah trees per site for five treatments representing different disturbance regimes. Trees were categorised into three different size classes; small (20 – 60cm d.o.b.), medium (60 - 100cm d.o.b.) or large (> 100cm d.o.b.). Different letters indicate significant differences (see Table 4.11).

Multivariate Analysis

Analysis of Similarity showed there were different attributes of tree and log sizes among the five treatments (Global R = 0.47, $p < 0.01$). All logged forest ('gap logged /burnt 2000', 'gap logged 1990' and 'selectively logged < 1950' treatments) contained different size class attributes than unlogged forest ('undisturbed' and 'burnt 2000' treatments), irrespective of fire history (Table 4.12). Undisturbed forest did not contain different size class attributes than forest burnt in 2000 (Table 4.12). Similarly, there were no differences in size classes among the three treatments that had been logged (Table 4.12).

Table 4.12 Summary of ANOSIM pairwise analysis testing whether the composition of attributes comprising different size classes of logs and trees were different between five treatments representing different disturbance regimes. Trees were categorised into three different size classes; small (20 - 60cm d.o.b.), medium (60 - 100cm d.o.b.) or large (> 100cm d.o.b.). Logs were categorised into three different size classes; small (0 - 0.1m^3), medium (0.1 - 1m^3) or large (> 1m^3). Highlighted rows denote pairs of treatments that were significantly different ($p < 0.05$).

Groups	**Global *R***	**P value**
Undisturbed, Burnt 2000	0.208	0.171
Undisturbed, Gap Logged/ Burnt 2000	0.75	0.029
Undisturbed, Gap Logged 1990	0.917	0.029
Undisturbed, Selectively Logged <1950	0.688	0.029
Burnt 2000, Gap Logged/ Burnt 2000	0.74	0.029
Burnt 2000, Gap Logged 1990	0.885	0.029
Burnt 2000, Selectively Logged <1950	0.594	0.029
Gap Logged/ Burnt 2000, Gap Logged 1990	0.042	0.343
Gap Logged/ Burnt 2000, Selectively Logged <1950	0.208	0.114
Gap Logged 1990, Selectively Logged <1950	0.135	0.200

There was a clear differentiation in the size classes of trees and logs between forest that has been disturbed by gap logging and unlogged forest on the ordination (Fig. 4.15). Sites in forest disturbed by gap logging in 1990 were more similar to forest disturbed by gap logging and fire in 2000 than sites in undisturbed forest or forest burnt in 2000 (Fig. 4.15). With the exception of one site representing forest burnt in 2000, sites in forest selectively logged prior prior to 1950 appeared to represent a boundary in size class attributes between forest disturbed by gap logging and unlogged forest (Fig. 4.15). On the ordination, sites in forest gap logged in 1990 and in forest selectively logged prior to 1950 also appeared more variable than sites in the other three treatments (Fig. 4.15).

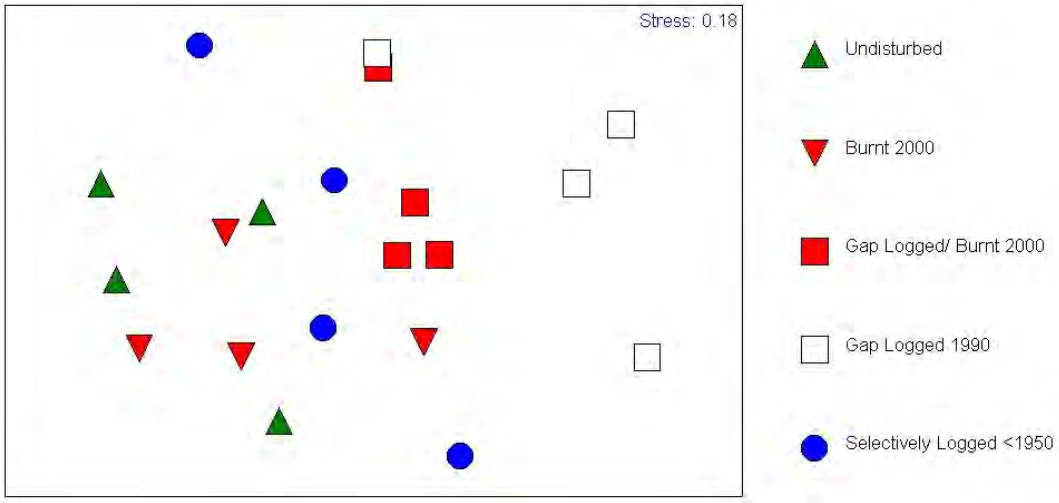


Fig 4.15 Non-metric multidimensional scaling plot showing the effect of five disturbance regimes on attributes that comprise different size classes of logs and trees. Trees were categorised into three different size classes; small (20 - 60cm d.o.b), medium (60 – 100cm d.o.b) or large (> 100cm d.o.b). Logs were also categorised into three different size classes; small (0 - 0.1m^3), medium (0.1 – 1m^3) or large (> 1m^3).

The attribute that contributed most to size class differences between unlogged treatments ('undisturbed' and 'burnt 2000') and logged treatments ('gap logged/burnt 2000', 'gap logged 1990', and 'forest selectively logged < 1950') was the volume of medium sized jarrah logs, which was greater in all previously logged forest than unlogged forests (Table 4.13). The greater SBA of medium sized jarrah trees in unlogged treatments compared to logged treatments also contributed to the differences between these treatments (Table 4.13). When size class attributes of logs and trees are compared between gap logged treatments ('gap logged/burnt 2000' and 'gap logged 1990' treatments) and unlogged treatments, the greater volume of large jarrah logs in gap logged treatments contributed to differences between these treatments (Table 4.13). In contrast, when size class attributes are compared between forest selectively logged prior to 1950 and both unlogged treatments, the greater volume of large jarrah logs in both the unlogged treatments contributed to differences between these treatments (Table 4.13).

Table 4.13 Size classes of trees and logs that contributed to differences between treatments identified as significant by pair-wise ANOSIM analysis (see Table 4.12). Attributes listed are those that contributed > 5 % of the difference found between treatments as determined by SIMPER analysis (percent contribution). Attributes are grouped within the treatment in which they had greater BA (m^2 ha^{-1}) or volume (m^3 ha^{-1}). Trees were categorised into three different size classes; small (20 - 60cm d.o.b.), medium (60 - 100cm d.o.b.) or large (> 100cm d.o.b.). Logs were categorised into three different size classes; small (0 - 0.1m^3), medium (0.1 - 1m^3) or large (> 1m^3).

Treatments	**Attributes contributing to difference (% of variation contributed by attribute)**
Undisturbed	SBA Medium Jarrah Trees (17.2)
Gap Logged/ Burnt 2000	Volume Medium Jarrah Logs (29.4) Volume Large Jarrah Logs (27.2)
Undisturbed	SBA Medium Jarrah Trees (10.3)
Gap Logged 1990	Volume Medium Jarrah Logs (26.6) Volume Large Jarrah Logs (17.1)
Undisturbed	Volume Large Jarrah Logs (28.2) SBA Medium Jarrah Trees (13.9)
Selectively Logged < 1950	Volume Medium Jarrah Logs (39.1) Volume Small Jarrah Logs (12.1)
Burnt 2000	SBA Medium Jarrah Trees (16.7)
Gap Logged/ Burnt 2000	Volume Medium Jarrah Logs (47.0) Volume Large Jarrah Logs (17.3)
Burnt 2000	SBA Medium Jarrah Trees (14.2)
Gap Logged 1990	Volume Medium Jarrah Logs (35.7) Volume Large Jarrah Logs (17.3)
Burnt 2000	Volume Large Jarrah Logs (17.3) SBA Medium Jarrah Trees (14.4)
Selectively Logged < 1950	Volume Medium Jarrah Logs (43.9) Volume Small Jarrah Logs (13.8)

4.4.4 Tree Cover

The codominant strata typically provided most canopy cover (between 36 - 95 % of total tree cover per site) followed by the subdominant vegetation (19 - 73 %) and dominant strata (5 - 14 %). Jarrah trees constituted most of the dominant (50 – 100 % of dominant cover per site) and codominant (52 - 99 %) cover. There

was greater variety in subdominant cover, with marri (0 - 35 % of subdominant cover per site), banksia (0 - 27 %) and sheoak (0 - 62.3 %) becoming an important component with jarrah (8 - 86 %). The 'other' component (*Persoonia longifolia, Persoonia elliptica, Xanthorrhoea preissii, Kingia australis* and *Nuytsia floribunda*) contributed between 0 – 37 % of the subdominant cover.

Univariate Analysis

All attributes except the structural diversity of tree cover were transformed using arcsine prior to analysis. Of the 15 attributes assessed, three (total canopy cover, total codominant cover and total subdominant cover) provided significantly different cover among treatments at standard levels of significance ($p < 0.05$; Table 4.14). Pairwise analysis showed that total canopy cover, total codominant cover and total subdominant cover was lower in forest gap logged and burnt in 2000 compared with the four other treatments ($p < 0.05$; Fig. 4.16).

Table 4.14 Summary of one-way ANOVAs for testing 15 attributes of tree cover (df = 19). Statistical power is also presented. Highlighted rows indicate those attributes that had significantly different tree cover among treatments (dark grey $p < 0.05$; light grey $p < 0.1$). '*' represents attributes that were transformed by natural log prior to conducting the ANOVA. Dashed lines differentiates subsets of attributes that are plotted on the same figure (see Fig. 4.16; Fig. 4.17; Fig. 4.18).

Attributes	*F*	P value	Observed Power
Total *	7.75	.01	0.97
Dominant*	2.60	.07	0.58
Codominant*	3.56	.03	0.72
Subdominant*	3.07	.04	0.66
Structural Diversity	0.63	.65	0.16
Dominant – Jarrah *	3.04	.06	0.66
Dominant – Marri *	3.07	.06	0.66
Codominant – Jarrah *	1.57	.23	0.37
Codominant – Marri *	2.10	.13	0.49
Codominant – Sheoak *	0.87	.51	0.21
Subdominant – Jarrah *	1.19	.36	0.29
Subdominant – Marri *	1.95	.16	0.45
Subdominant – Banksia *	2.92	.06	0.74
Subdominant – Sheoak *	2.31	.11	0.53
Subdominant – Other *	0.60	.67	0.16

When alpha was adjusted to account for low statistical power, four additional attributes had different cover values between treatments ($p < 0.1$; Table 4.14). These attributes were dominant cover of all trees, the dominant cover of jarrah, dominant cover of marri and subdominant cover of banksias. Pairwise analysis showed there was less cover provided by dominant trees in forest gap logged and burnt in 2000 than unlogged forest, forest selectively logged prior to 1950 and forest burnt in 2000 ($p < 0.1$; Fig. 4.16).

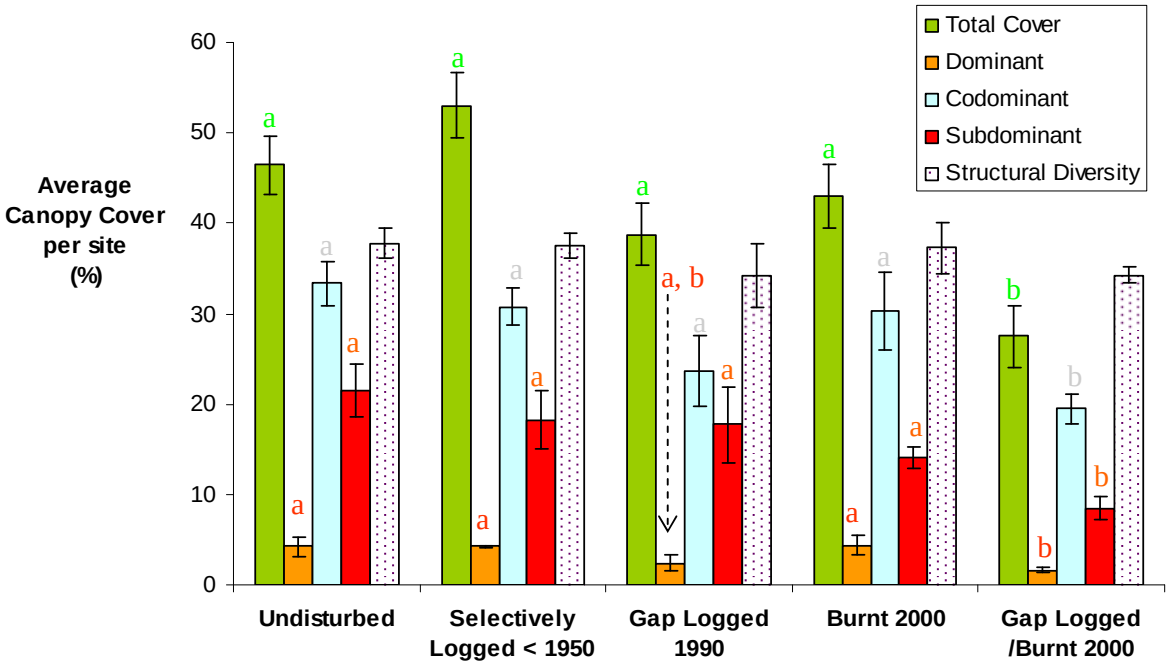


Fig. 4.16 The average (± SE) total cover, dominant cover, codominant cover and subdominant cover for trees > 2m in height per site for five treatments representing different disturbance regimes. The structural diversity (Equation 4.6) of canopy is also represented as a percentage (i.e, Diversity Index x 100). Different letters indicate significant differences (see Table 4.14).

The cover provided by dominant jarrah and marri was different among treatments when alpha was adjusted to account for low statistical power ($p < 0.1$; Fig. 4.17). Pairwise analysis showed the cover provided by dominant jarrah trees is higher in undisturbed forest, forest selectively logged prior to 1950 and forest burnt in 2000 than in forest disturbed by gap logging ('gap logged 1990' and 'gap logged/ burnt 2000' treatments ($p < 0.1$; Fig. 4.17). Furthermore, the cover provided by dominant marri trees is higher in the three treatments not burnt in 2000 ('undisturbed', 'forest selectively logged < 1950' and 'gap logged 1990') than in both treatments that were burnt in 2000 ('burnt 2000' and 'gap logged burnt 2000') ($p < 0.1$; Fig. 4.17).

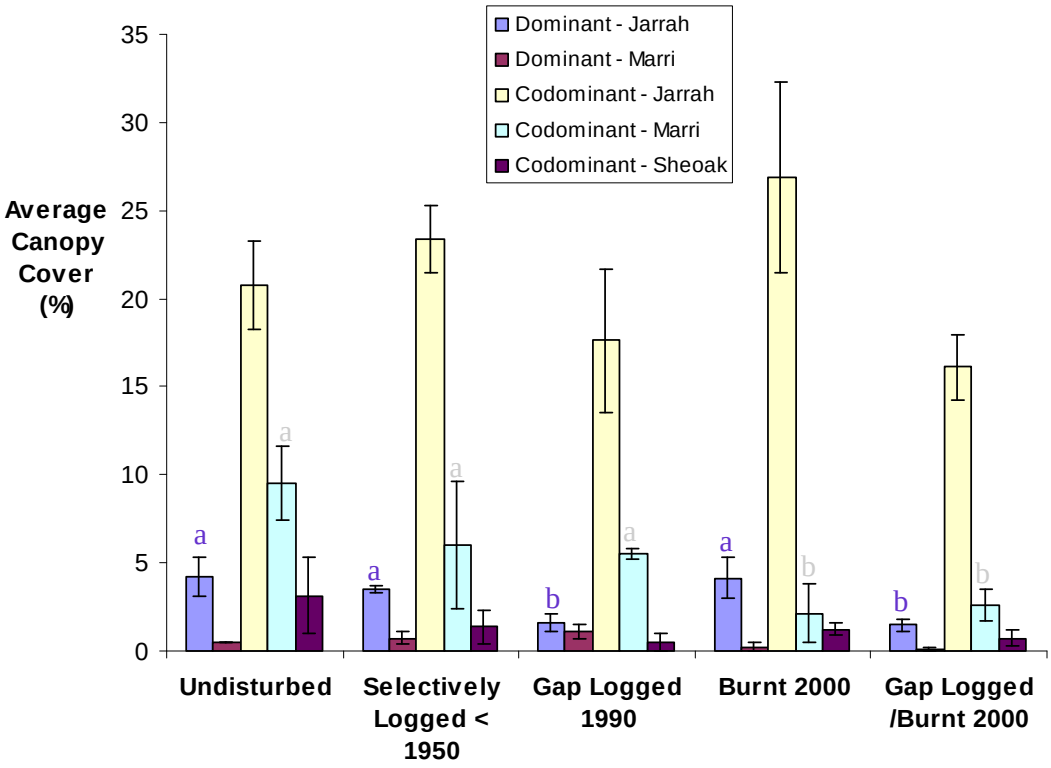


Fig. 4.17 The average (± SE) cover of dominant jarrah and marri, and codominant jarrah, marri and sheoak for trees > 2m in height per site for five treatments representing different disturbance regimes. Different letters indicate significant differences (see Table 4.14).

The cover provided by subdominant banksia was different among treatments when alpha was adjusted to account for low statistical power ($p < 0.1$). Pairwise analysis showed there was less cover by subdominant banksia in both treatments that were gap logged ('gap logged 1990' and 'gap logged / burnt 2000') compared to the three other treatments ($p < 0.1$; Fig. 4.18). Furthermore, the cover provided by subdominant jarrah trees in forest gap logged in 1990 was higher than in undisturbed forest, forest burnt in 2000, and forest gap logged and burnt in 2000 ($p < 0.1$; Fig. 4.18). There was no difference in the cover provided by subdominant jarrah trees between forest selectively logged prior to 1950 and all other treatments ($p > 0.1$; Fig. 4.18).

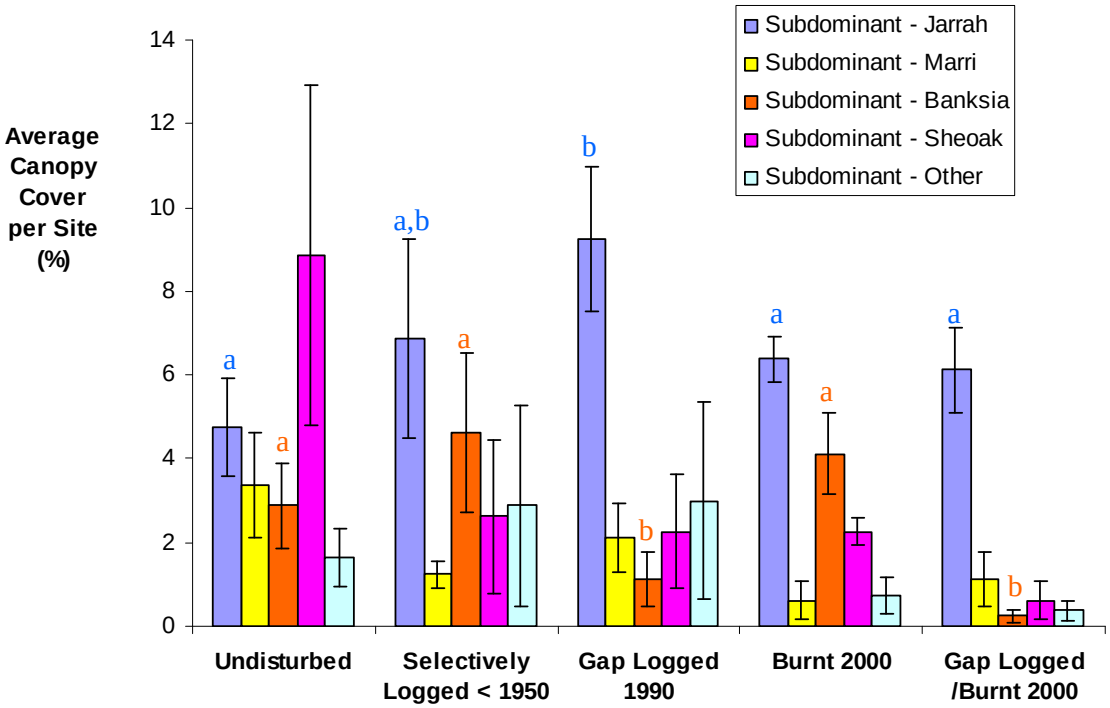


Fig. 4.18 The average (± SE) cover of subdominant jarrah, marri, sheoak, banksia and 'other' trees > 2m in height per site for five treatments representing different disturbance regimes. Different letters indicate significant differences (see Table 4.14).

Multivariate Analysis

Four attributes of tree cover were removed (total canopy, total dominant cover, total codominant cover, total subdominant cover, and canopy structural diversity) because they were collinear with other attributes (correlation coefficient > 0.8). Analysis of Similarity of the remaining 10 attributes showed there were significant differences in tree cover attributes among treatments (Global R = 0.226, $p < 0.01$). Pairwise analysis showed sites that had been gap logged and burnt in 2000 had different attributes of canopy cover than undisturbed forest and forest selectively logged prior to 1950 (Table 4.15). This analysis also showed that attributes of canopy cover in forest gap logged and burnt in 2000 were approaching statistical significance when compared with forest burnt in 2000 ($p = 0.057$) and forest gap logged in 1990 ($p = 0.057$) (Table 4.15). Sites in forest gap logged and burnt (2000) were less variable in attributes of tree cover than the

other four treatments on the ordination (Fig. 4.19). Sites in the other four treatments showed a high degree of heterogeneity with respect tree cover attributes, and there appeared to be no delineation in sites due disturbance regimes (fig. 4.19).

Table 4.15 Summary of ANOSIM pairwise analysis testing whether the composition of attributes that comprised tree cover was different between five treatments representing different disturbance regimes. Highlighted rows denote pairs of treatments that were significantly different ($p < 0.05$).

Groups	**Global *R***	**P value**
Undisturbed, Burnt 2000	0.229	0.114
Undisturbed, Gap Logged/ Burnt 2000	0.552	0.029
Undisturbed, Gap Logged 1990	0.26	0.114
Undisturbed, Selectively Logged <1950	0.031	0.443
Burnt 2000, Gap Logged/ Burnt 2000	0.521	0.057
Burnt 2000, Gap Logged 1990	0.219	0.143
Burnt 2000, Selectively Logged <1950	-0.125	0.714
Gap Logged/ Burnt 2000, Gap Logged 1990	0.24	0.057
Gap Logged/ Burnt 2000, Selectively Logged <1950	0.49	0.029
Gap Logged 1990, Selectively Logged <1950	0.01	0.50

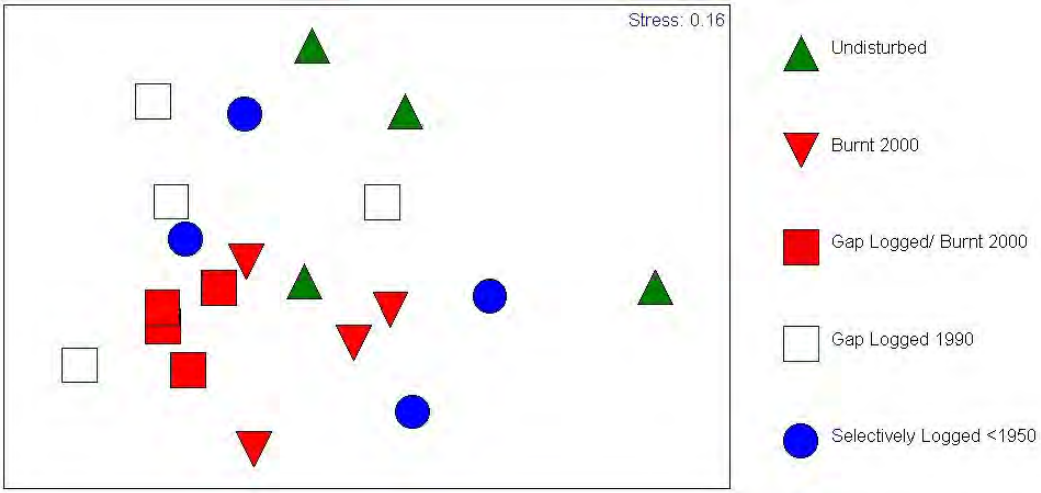


Fig. 4.19 Non-metric multidimensional scaling plot showing the effect of five disturbance regimes on attributes that comprise tree cover.

SIMPER analysis showed that the greater cover of jarrah codominant trees in undisturbed forest and forest selectively logged prior to 1950 was the attribute that contributed most to differences between these treatments, and forest gap logged and burnt in 2000 (Table 4.16). The greater cover of marri codominant and total subdominant cover in undisturbed forest and forest selectively prior to 1950 also contributed to the differences found among these three treatments (Table 4.16). Additionally, forest gap logged and burnt in 2000 had less sheoak cover than undisturbed forest, and less jarrah subdominant and banksia subdominant cover than forest selectively logged prior to 1950 (Table 4.16).

Table 4.16 Attributes of tree cover that contributed to differences found between treatments identified as significant by pair-wise ANOSIM analysis (see Table 4.14). Attributes listed are those that contributed greater than 5% of the difference found between treatments as determined by SIMPER analysis. Attributes are grouped within the 'treatment' in which they had greater cover (%), and ranked according to their contribution to difference between treatments.

Treatments	**Attributes contributing to difference (% of variation contributed by attribute)**
Undisturbed	Jarrah Codominant (17.9) Sheoak Subdominant (13.5) Marri Codominant (10.1) Total Subdominant (9.2)
Gap logged / Burnt 2000	-
Forest logged selectively <1950	Jarrah Codominant (17.1) Marri Codominant (12.9) Jarrah Subdominant (12.6) Total Subdominant (10.4) Banksia Subdominant (9.7)
Gap logged /Burnt 2000	-

4.4.5 Ground cover

Most vegetation cover occurred between 0 - 0.2 m off the ground for all sites (63 - 83 % of vegetation cover at each site). The vegetation cover between 0.2 - 0.4 m (22 - 78 %), 0.4 - 1 m (10 – 88 %) and 1-2 m (0 – 15 %) was variable. Other surface interface attributes varied between sites, and consisted primarily of leaf cover (14 - 91 % of ground cover), bare soil (4 - 77 %) or woody debris (1 – 13 %). Dead stumps (0 - 3 %) and other attributes (ashbeds, moss, termite mounds, stump holes) (0 - 2 %) were relatively uncommon at all sites.

Univariate

All ground attributes except structural diversity were transformed using arcsine prior to statistical testing by analysis of variance. Six of the 12 attributes were significantly different among treatments at standard levels of significance ($p < 0.05$; Table 4.17). These six attributes were understorey cover between 0.2 - 0.4m and between 0.4 – 1m, leaf cover, log cover, bare soil cover and stump cover. When alpha was adjusted to account for low statistical power (i.e., power < 0.8), the ground structural diversity was different among treatments ($p < 0.1$).

Table 4.17 Summary of one-way ANOVAs for 12 attributes of ground cover (df = 19). Statistical power is also presented. Highlighted rows indicate those attributes that had significantly different cover between treatments representing different disturbance regimes (dark grey $p < 0.05$; light grey $p < 0.1$). '*' represents attributes that were transformed by Arcsin prior to statistical testing.

Attributes	*F*	P Value	Observed Power
Total Shrub Cover *	1.54	.24	0.36
Cover 0 – 0.2m *	1.04	.42	0.25
Cover 0.2 – 0.4m *	8.44	.00	0.99
Cover 0.4 – 1m *	5.66	.01	0.92
Cover 1 – 2m *	1.52	.25	0.36
Structural Diversity	3.34	.06	0.71
Leaf Cover *	158.32	.00	1.0
Rock Cover *	1.72	.20	0.41
Log Cover *	18.13	.00	1.0
Other Cover *	0.41	.80	0.12
Bare Soil Cover *	147.98	.00	1.00
Stump Cover *	12.21	.00	0.99

Pairwise analysis showed the cover of vegetation between 0.2 - 0.4 m and 0.4 – 1.0 m was lower in recently burnt sites ('burnt 2000' and 'gap logged/burnt 2000' treatments) compared to sites in the three other treatments ($p < 0.05$ Fig. 4.20). There was also lower structural diversity in both the treatments burnt in 2000 than the other three treatments (Fig. 4.20). The vegetation cover between 0 - 0.2 m and 1 – 2 m was similar among all treatments ($p > 0.05$; Fig 4.20).

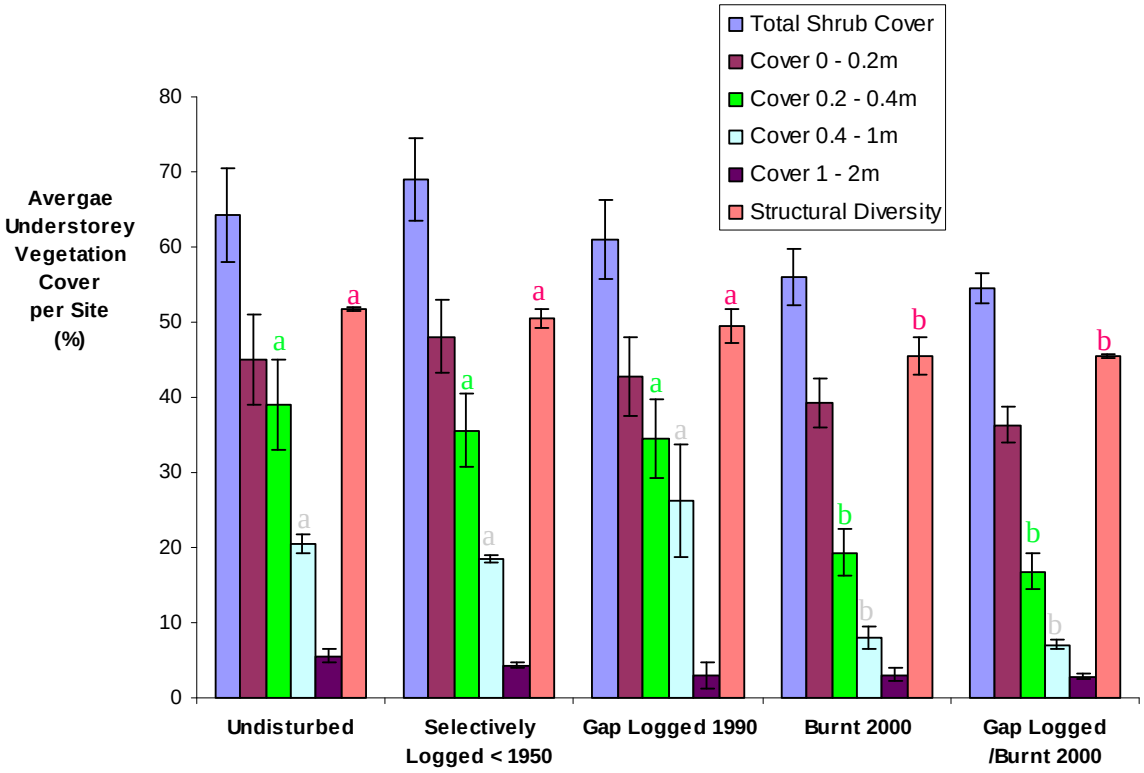


Fig. 4.20 The average (± SE) cover of understorey vegetation per site for five treatments representing different disturbance regimes. Four strata of vegetation cover were assessed; 0 – 0.2m height above ground, 0.2 – 0.4m, 0.4 – 1m, and 1 – 2m. In addition, the ground vegetation structural diversity (Equation 4.6) is also presented as a percentage (i.e., structural diversity index x 100). Different letters indicate significant differences (see Table 4.17).

Pairwise analysis showed the cover of leaf litter was lower in recently burnt sites compared with forest that had not been burnt for five years ($p < 0.05$; Fig. 4.21). In contrast, the cover of bare sand was higher in sites that were burnt in 2000 ($p < 0.05$; Fig. 4.21). The ground cover of logs was also higher in the two treatments disturbed by gap logging ('gap logged/burnt 2000' and 'gap logged 1990') than

the other three treatments ($p < 0.05$; Fig. 4.21). Furthermore, the cover provided by the stumps was higher in the three treatments that had been disturbed by logging ('gap logged/burnt 2000', 'gap logged 1990' and 'selectively logged < 1950') than forest that had never been logged ($p < 0.05$; Fig. 4.21).

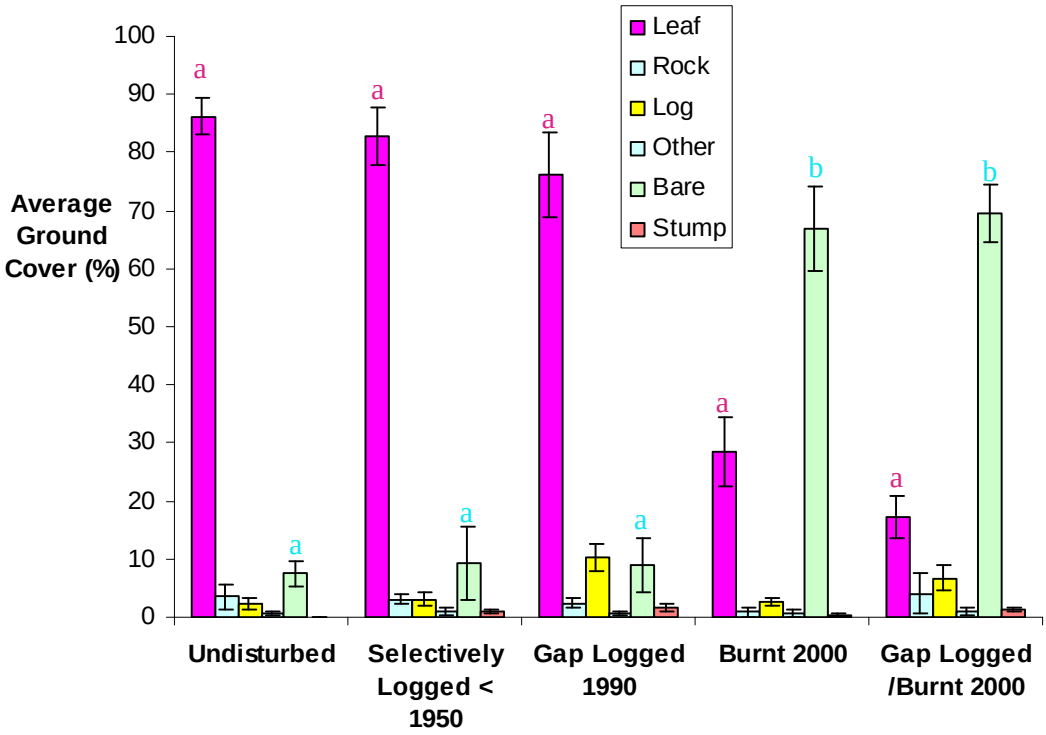


Fig. 4.21 The average (± SE) cover of non-vegetation attributes per site for five treatments representing different disturbance regimes. Different letters indicate significant differences (see Table 4.17).

Multivariate

Three variables were collinear with attributes (correlation coefficient > 0.8) and removed from further analysis (total shrub cover, bare soil and understorey structural diversity). Multivariate analysis of the remaining nine variables showed there were significant differences in attributes of ground cover among the five treatments (Global $R = 0.548$, $p < 0.01$). Pairwise analysis showed all sites that had been burnt in 2000 (irrespective of logging) were different from all sites that had not been burnt for at least five years (Table 4.18).

Table 4.18 Summary of ANOSIM pairwise analysis testing whether the composition of attributes that comprised ground cover were different between five treatments representing different disturbance regimes. Highlighted rows denote pairs of treatments that were significantly different.

Groups	Global *R*	P value
Undisturbed, Burnt 2000	0.781	0.029
Undisturbed, Gap Logged/ Burnt 2000	0.875	0.029
Undisturbed, Gap Logged 1990	0.615	0.029
Undisturbed, Selectively Logged <1950	0.146	0.286
Burnt 2000, Gap Logged/ Burnt 2000	0.24	0.171
Burnt 2000, Gap Logged 1990	0.74	0.029
Burnt 2000, Selectively Logged <1950	0.823	0.029
Gap Logged/ Burnt 2000, Gap Logged 1990	0.552	0.029
Gap Logged/ Burnt 2000, Selectively Logged <1950	0.75	0.029
Gap Logged 1990, Selectively Logged <1950	0.26	0.057

There were clear differences in ground attributes between the five treatments on the ordination (Fig. 4.22). Sites in that were burnt in 2000 are separated from unburnt sites (Fig. 4.22). Sites in forest gap logged in 1990 were the most variable (Fig. 4.22). Sites found in forest selectively logged prior to 1950 were between undisturbed sites and sites located in forest gap logged in 1990 (Fig. 4.22). The least variable sites were those found in unlogged forest, forest burnt in 2000 and forest selectively logged < 1950 (Fig. 4.22).

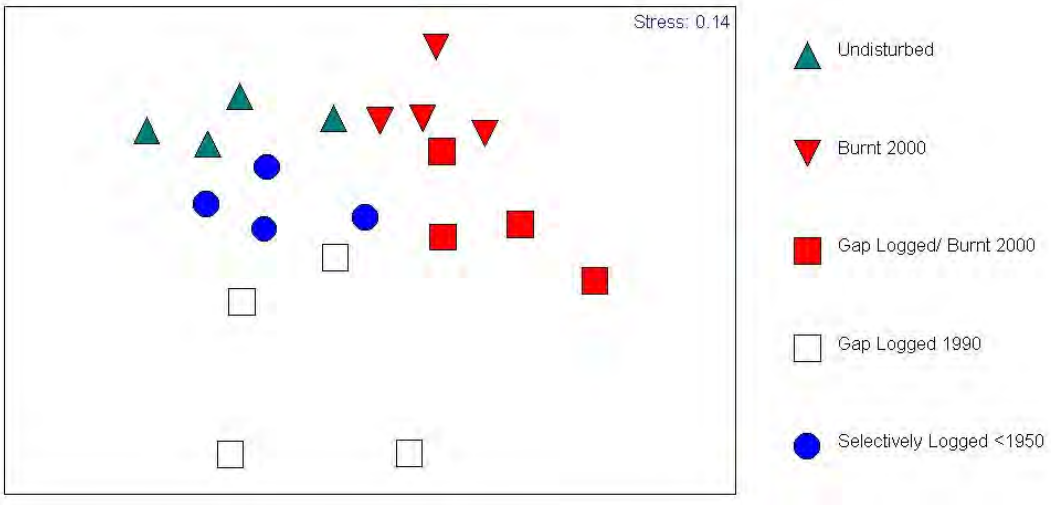


Fig. 4.22 Non-metric multidimensional scaling plot showing the effect of five disturbance regimes on attributes that comprise ground cover.

SIMPER analysis showed that greater leaf litter cover and vegetation cover between 0.2 – 0.4 m in the three treatments that were not burnt in 2000 (‘undisturbed’, ‘gap logged 1990’ and ‘selectively logged < 1950) contributed most of the differences between these treatments and forest that were burnt in 2000 (‘burnt 2000’ and ‘gap logged burnt 2000’). Furthermore, higher log and branch cover in forest that had been gap logged (1990 & 2000) compared to undisturbed forest, burnt (2000) and forest logged < 1950 contributed to differences between these treatments (Table 4.18; Table 4.19). Moreover, the greater cover of plants between 0.4 - 1m in forest gap logged in 1990 contributed to differences between this treatments and undisturbed forest, forest gap logged and burnt in 2000 and forest burnt in 2000 (Table 4.19).

Table 4.19 Attributes of ground cover that contributed to differences found between treatments identified as significant by pair-wise ANOSIM analysis (see Table 4.18). Attributes listed are those that contributed greater than 5% of the difference found between treatments as determined by SIMPER analysis. Attributes are grouped within the treatment in which they had greater cover (%), and ranked according to their contribution to difference between treatments.

Treatments	**Attributes contributing to difference (% of variation contributed by attribute)**
Gap Logged/ Burnt 2000	Log and Branch (9.4)
Undisturbed	Leaf Litter (40.2) Vegetation 0.2 - 0.4m (26.7) Vegetation 0.4 – 1m (7.1)
Gap Logged/ Burnt 2000	-
Gap Logged 1990	Leaf Litter (38.7) Vegetation 0.2 - 0.4m (30.9) Vegetation 0.4 – 1m (8.8)
Gap logged/ Burnt 2000	Log and Branch (9.5)
Selectively Logged < 1950	Leaf Litter (35.5) Vegetation 0.2 - 0.4m (26.8) Vegetation 0.4 – 1m (7.6)
Burnt 2000	-
Undisturbed	Leaf Litter (39.1) Vegetation 0.2 - 0.4m (36.2) Vegetation 0.4 – 1m (10.2)
Burnt 2000	-
Gap Logged 1990	Vegetation 0.2 - 0.4m (36.7) Leaf Litter (28.5) Vegetation 0.4 - 1m (11.6) Log and Branch (10.6)
Burnt 2000	-
Selectively logged < 1950	Leaf Litter (40.2) Vegetation 0.2 - 0.4m (34.3) Vegetation 0.4 - 1m (11.7)
Undisturbed	Leaf Litter (26.4)
Gap Logged 1990	Vegetation 0.4 - 1m (18.3) Log and Branch (15.9)

4.4 Discussion

Logging disturbs jarrah forest structure for a greater length of time than fire. Other studies in jarrah forest have shown structural attributes affected by fire (e.g., leaf litter and understorey vegetation) recover within 5 - 7 years (Bell ***et al.*** 1989; Christensen & Abbott 1989; Van Huerck ***et al.*** 1998). In this research, structural changes caused by selective logging were discernible after 50 years, with smaller trees and greater volumes of CWD recorded in logged forest compared with unlogged forest. Although there was no 50 year-old gap logged sites for comparison, gap logging appears to exacerbate these structural changes, producing stands with fewer trees and larger volumes of CWD than either unlogged forest or forest selectively logged prior to 1950. The slow growth rate of jarrah (Abbott & Loneragan 1986; O'Connell & Mendham 2004), and slow decomposition of CWD (Brown ***et al.*** 1996), suggests structural changes will exist for at least two centuries after logging.

The structure of unlogged forest is an outcome of disturbance, competition, and site –specific properties. This research showed that structural characteristics associated with trees (e.g., stand basal area, the cover of small, medium and large trees of all species, and volume and composition of CWD) varied considerably between unlogged sites. Although significant variation in tree composition, cover and size occurs because of edaphic reasons (Abbott & Loneragan 1983a; Stoneman ***et al.*** 1995), the multiple size and age structure of trees in unlogged forest is also the result of subdominant trees competing in response to the weakening of the overstorey as old trees enter senescence (Abbott 1984b; Abbott & Loneragan 1986). The age of the trees within an unlogged patch directly influences the composition and decay of CWD. This is because as trees age, they

produce a continual supply of well-decayed woody debris in the form of sticks, branches and eventually boles (Lindenmayer 2003). Given that unlogged jarrah forests are typically dominated by mature and over-mature trees (Abbott & Loneragan 1986), the majority of CWD is relatively well-decayed.

4.4.1 The Impact of Fire in Unlogged and Logged Forest

Fire affected several structural attributes. There was less cover provided by leaf litter and understorey vegetation 0.2m to 1m from the ground in both treatments disturbed by fire in 2000 (unlogged/burnt 2000 and gap logged/burnt 2000) compared to the three other treatments. Other studies in jarrah forest have shown these attributes can take up to 7 years to recover after fire (e.g., Hingston 1985; Bell *et al.* 1989; Christensen & Abbott 1989; Van Huerck *et al.* 1998). In contrast to these highly flammable ground attributes, the fire at Mt Dale did not appear to influence the stand basal area or canopy cover of trees. This is because the bark of mature jarrah and marri trees is thick and provides a degree of protection from heat during fire (Attiwill 1994). Only intense summer fires are able to kill large these mature trees (Abbott & Loneragan 1986; Attiwill 1994). Furthermore, the 2000 fire at Mt Dale did not appear to reduce the volume of CWD. Although fire did burn the outside of logs, stags and stumps (pers. observ.), and has been shown to hasten some aspects of the decay process (e.g., fire helps form hollows) (Williams & Faunt 1997), most of the CWD remained this fire.

There were substantial structural differences between unlogged/burnt forest and gap logged/burnt forest. At the gap logged sites, the cutting down of jarrah trees (typically medium sized, dominant, decay class 2 trees), and the creation of gaps,

significantly reduced stand basal area and canopy cover attributes. Furthermore, there were much larger volumes of CWD (typically decay class 1 and 2 jarrah logs, but also balga) in the gap logged/burnt forest than the unlogged/burnt forest. This is the result of current managers only removing the economically important component of the jarrah tree (the bole) off-site. Dead stumps, 'tops' (the portion of tree that consists of the crown above the bole) and pushed over understorey (mainly sheoak, banksias, balga and marri) are left on site, and add to the volume CWD that already existed within these ecosystems (Fig. 4.23).

Fig. 4. 23 Photo showing the volume of coarse woody debris after gap logging.

4.4.2 Stand Structure 10 years after Gap Logging

Forest that was gap logged in 1990 appeared to have recovered from the effects of fire. This is because the cover provided by leaf litter and understorey vegetation cover were equivalent in forest gap logged in 1990 and unlogged forest. In contrast, forest that had been gap logged in 1990 had not recovered from logging disturbance. As with forest gap logged in 2000, forest that was gap logged in 1990 had lower stand basal area (typically medium sized, dominant, decay class 2 jarrah trees) than unlogged forest. Furthermore, treatments that

were gap logged in 1990 also had much larger volumes of CWD (typically decay class 1 and 2 jarrah logs) than unlogged treatments.

The similarity in stand basal area of tree attributes and the volume of CWD attributes between forest gap logged in 1990 and forest gap logged in 2000 reveals that recovery from gap logging disturbance is slow. Estimates using Whitford's (2002) relationship between tree age and d.o.b. (tree age = 2.345 x d.o.b. + 6.968, R^2= 0.82, n=99 jarrah trees) suggest it will take at least 240 years for seedlings to reach the 100cm d.o.b (large status). As gap logging is effectively a stand replacing disturbance (Burrows *et al.* 2002a), it is only when seedlings reach large status that forest size structure is likely to be equivalent to unlogged forest. Therefore, it is predicted that it will take at least 240 years before a gap logged patch contains the size-structure found in unlogged forest (Fig. 4.24). It must be noted that although Whitford's (2002) formulae is the most rigorous estimate of the relationship between tree age and basal area, my estimations should be interpreted with some caution as site specific characteristics will affect tree growth (Abbott & Loneragan 1983a; Stoneman *et al.* 1995). This period of time is also likely to be an underestimate because subdominant trees can remain dormant for centuries before they have the opportunity to grow and become codominant or dominant trees (Bradshaw 2002). After logging, it may take several generations of the dominant overstorey to recover the full range of attributes found in unlogged forest (e.g., large live trees, large stags, collapsed trees in various stages of decay) (Norton 1996). As the current rotation lengths for the majority of the gap logged forest is 170 years (ranging from 100-220

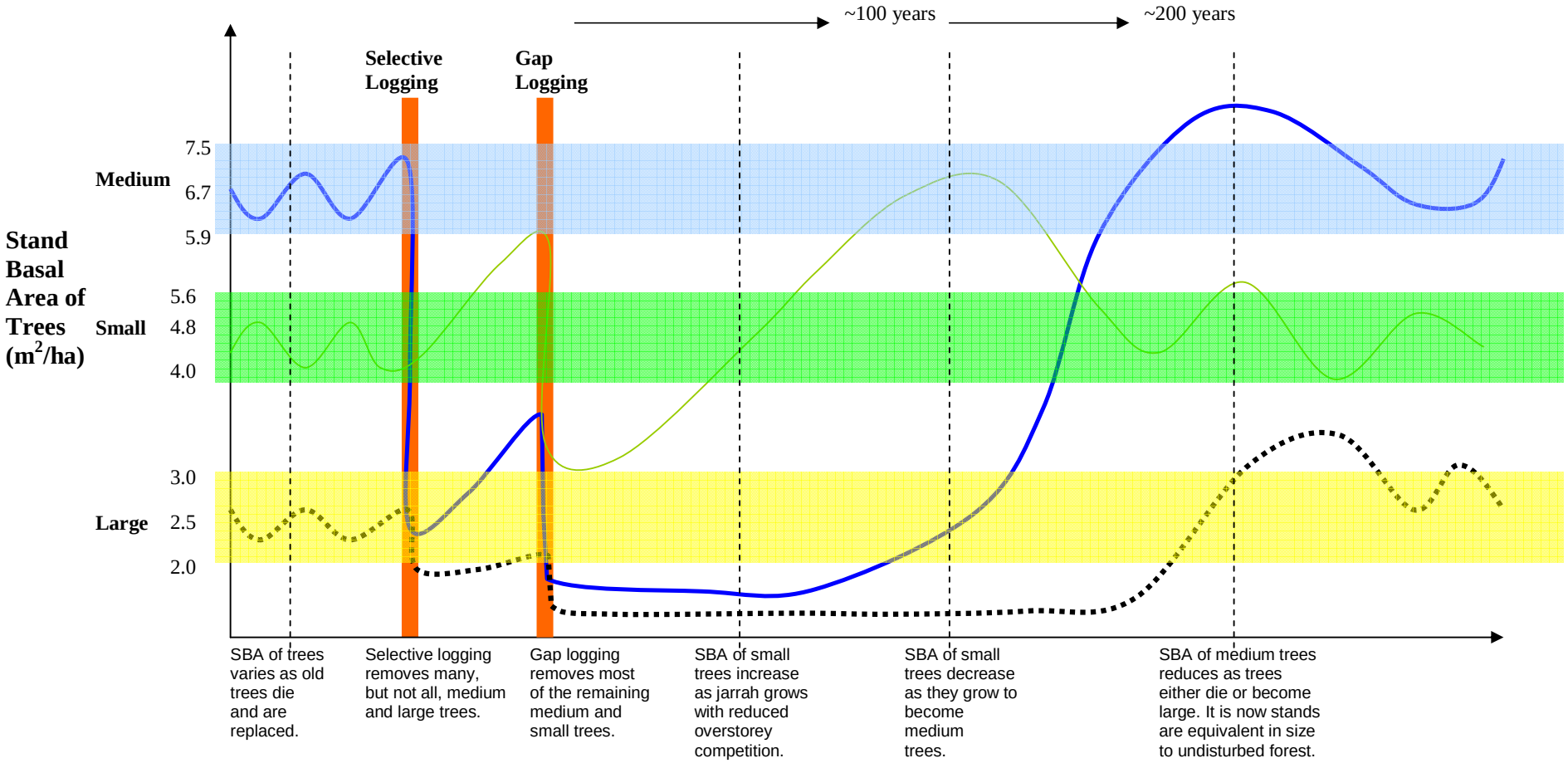


Fig. 4.24 A conceptual model of change in stand basal area (SBA) of at Mt Dale if unlogged forests were selectively logged, and then gap logged 40 years later. Trees are delineated as 'small' (20 - 60cm d.o.b.), 'medium' (60 - 100 cm d.o.b.) and 'large' (> 100cm d.o.b.). Age-size estimates are calculated using age = 2.345 x d.o.b. + 6.968 (Whitford 2002). Shaded regions represent the standard error (± SE) of the mean SBA for each size class of jarrah trees in unlogged forest. This model is likely to underestimate the time taken for recovery as it assumes that recovery comes from subdominant trees that are all 20cm d.o.b. at the time of logging. It has been shown that regeneration of jarrah after logging comes from smaller plants including existing coppice, saplings, and poles (Crombie 1997). Furthermore, this model does not account for dormancy in growth caused by competing subdominant trees (Bradshaw 2002).

years) (see Ferguson *et al.* 2003), it is unlikely that logged forests will ever attain the size structure of trees or CWD currently found in unlogged forest.

The volume of CWD in forest gap logged in 1990 is similar to forest gap logged in 2000, and therefore is much greater to that found in unlogged forest. The CWD in gap logged forest comprises the logging residue as well as well decayed logs that naturally exist in the ecosystem. Hagan and Grove (1999) predicted it can take centuries for CWD to decompose in temperate forests. As Brown *et al.* (1996) showed that small jarrah logs (10 cm d.o.b.) decompose more slowly than CWD in other ecosystems, changes in the volume of CWD resulting from gap logging could be expected to exist for decades if not centuries based on Hagan and Grove's prediction.

Changes in forest structure following logging are expected to have negative and positive effects on components of jarrah biodiversity. The reduction in stand basal area of mature trees following logging could have deleterious consequences for species that require mature trees (e.g., nesting cockatoos, bats, marsupials) (e.g., Saunders 1978; Lindenmayer 1992; Zielinski & Gellman 1998). In contrast, the increase in structural heterogeneity provided by coarse woody debris could increase the saprotrophic species within an area (McGee *et al.* 1999). Research is required to assess what impact the decrease in mature trees and increase in CWD has on jarrah forest biodiversity. Modelling tree growth, death and decay is required to accurately determine the time required for forests to fully recover from contemporary logging disturbance (see Gibbons & Lindenmayer 1996; Grove *et al.* 2002). Moreover, assessing what impact the removal of live wood influences nutrient cycling is also required to understand the long-term

sustainability of logging. Given that a large amount of wood is now removed as a result of contemporary logging practices, a substantial amount of the nutrient pool may be exported out of the jarrah forest as a result of this disturbance (Hingston *et al.* 1980; Hingston *et al.* 1989).

Although most tree structure and CWD attributes were similar between forest gap logged in 1990 and forest gap logged / burnt in 2000, there was lower stand basal area of decay 4 jarrah trees in forest that was gap logged in 1990 than in forest gap logged/burnt in 2000. Other studies have shown that after logging, remaining trees have a propensity to fall over, possibly as a result of changes in exposure to environmental stresses including storms (Lindenmayer *et al.* 1990). The lower stand basal area of decay 4 trees in forest gap logged in 1990 could indicate that these trees have fallen since sites were logged in 1990. However, this is unlikely as approximately 2% of 'habitat trees' (mature trees that are conserved in logging coupes to provide habitat to hollow bearing species) in jarrah forest fall per decade (Whitford & Williams 2001). Differences in the stand basal area of decay 4 trees between the two gap logged treatments is more likely to reflect the changes in management practices in the last decade, with greater emphasis being placed on preserving trees for habitat.

The vegetation cover in forest gap logged in 1990 was equivalent to unlogged forest. This supports other research that showed that 80% of the canopy cover (as projected from the ground) can recover in 10 years (Stoneman *et al.* 1989a). Several sites in forest gap-logged in 1990 also had a much greater cover of vegetation between 0.4 - 1m than that other treatments. This increase is primarily due to *Dryandra sessilis*, a species that thrives in recently disturbed areas (Beard

1990). It is expected that the cover between 0.4 - 1m will reduce as *D. sessilis* is replaced by overstorey tree species.

4.4.3 Forest Structure in Forest Selectively Logged Prior 1950

Changes in structure resulting from disturbance by selective logging was evident fifty years after the event. Differences between previously selectively logged forest and unlogged forest include much larger volumes of CWD in selectively logged forest compared to unlogged forest. This CWD comprised dead stumps, medium jarrah logs, and to a lesser degree small jarrah logs. With respect to decay, selectively logged forest had more jarrah logs of decay class 3 and decay 2 than unlogged forest. The stand basal area of medium sized jarrah trees (predominately decay class 2) and large jarrah logs also contributed to differences between these two treatments.

Differences in size-structure of trees between unlogged and selectively logged forest indicate that recovery from selective logging may take centuries. Whitford's (2002) equation estimates it takes between 90-100 years for a small tree (20cm) to reach a tree in the medium category (60cm), and a further 100 years before these medium sized trees become large. It is only when small trees reach large status that forest size structure is likely to be equivalent to pre-logged conditions (Fig. 4.24). Therefore, it is predicted that it will take at least 200 years before a selectively cutover patch contains the size-structure found in unlogged forest.

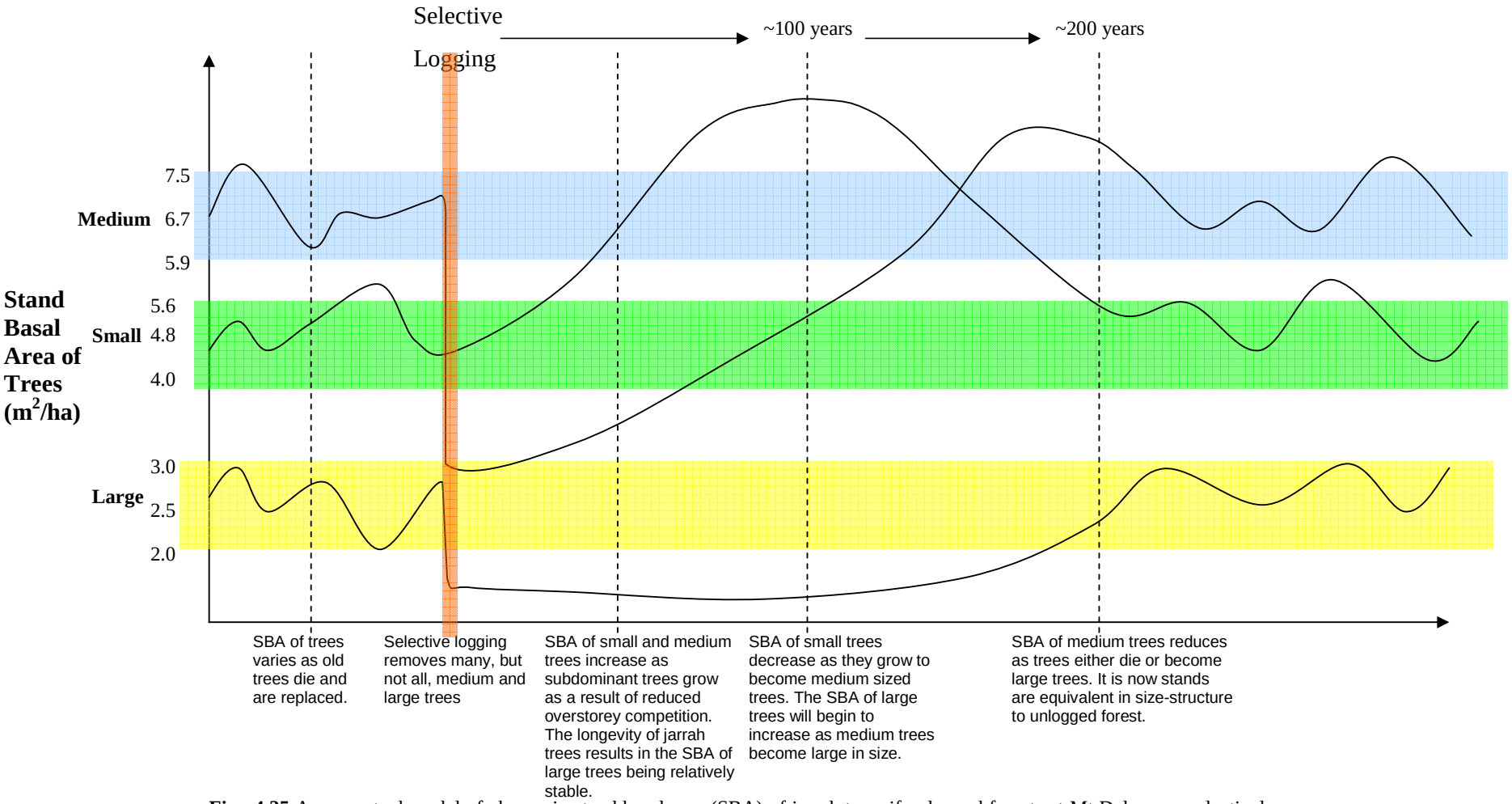


Fig. 4.25 A conceptual model of change in stand basal area (SBA) of jarrah trees if unlogged forests at Mt Dale were selectively logged. Trees are delineated as 'small' (20 - 60cm d.o.b.), 'medium' (60 - 100cm d.o.b) and 'large' (> 100cm d.o.b). Shaded regions represent the standard error (± SE) of the mean SBA for each size class of jarrah trees in unlogged forest**.** Age estimates are calculated from Whitford (2002). This model is likely to underestimate the time taken for recovery as it assumes that recovery comes from subdominant trees that are all 20cm d.o.b. at the time of logging. It has been shown that regeneration of jarrah after logging comes from smaller plants including existing coppice, saplings, and poles (Crombie 1997). Furthermore, this model does not account for dormancy in growth that result from competing subdominant trees (Bradshaw 2002).

Studying forest that was selectively logged prior to 1950 allows for an investigation of the dynamics of jarrah decay. Although selectively logged forest does contain a similar stand BA of over-mature trees (decay 4 jarrah trees) to unlogged forest, reductions in the stand BA of a younger cohort (decay 2 jarrah trees) is likely to cause a reduction in over-mature trees sometime in the future. This reduction in decay 2 jarrah trees is the result of past logging that removed economically viable trees (i.e., most of the older trees contained imperfections because of decay). Forest disturbed by selective logging can only be expected to contain a similar decay structure to unlogged forest when the current stand of young trees (decay 1) enters senescence. The length of time it takes jarrah to decay unknown, although the estimated age of this species (200-1400+ years) suggest this period of time could be a substantial (Mawson & Long 1994; Stoneman *et al.* 1997; Mawson & Long 1997). This study therefore supports the assertion that selective logging has produced a younger forest (Abbott & Loneragan 1986; Calver & Wardell-Johnson 2004).

The larger volumes of CWD in selectively logged forest is mostly the result of logging debris. This study found that most of the CWD in selectively logged forest was typically decay 3 and 4 jarrah trees, which indicates it takes at least 50 years for decay 1 logs to become decay 3 logs. This is because most of the CWD in these forest were once the 'tops' left behind after logging (and therefore, would have been decay 1 logs given only vigorous, healthy trees were selected to be logged). Despite these signs of decay, the larger volume of CWD in forest selectively logged prior to 1950 shows that it will take a significant amount of time for the volume of CWD to become equivalent to unlogged forest.

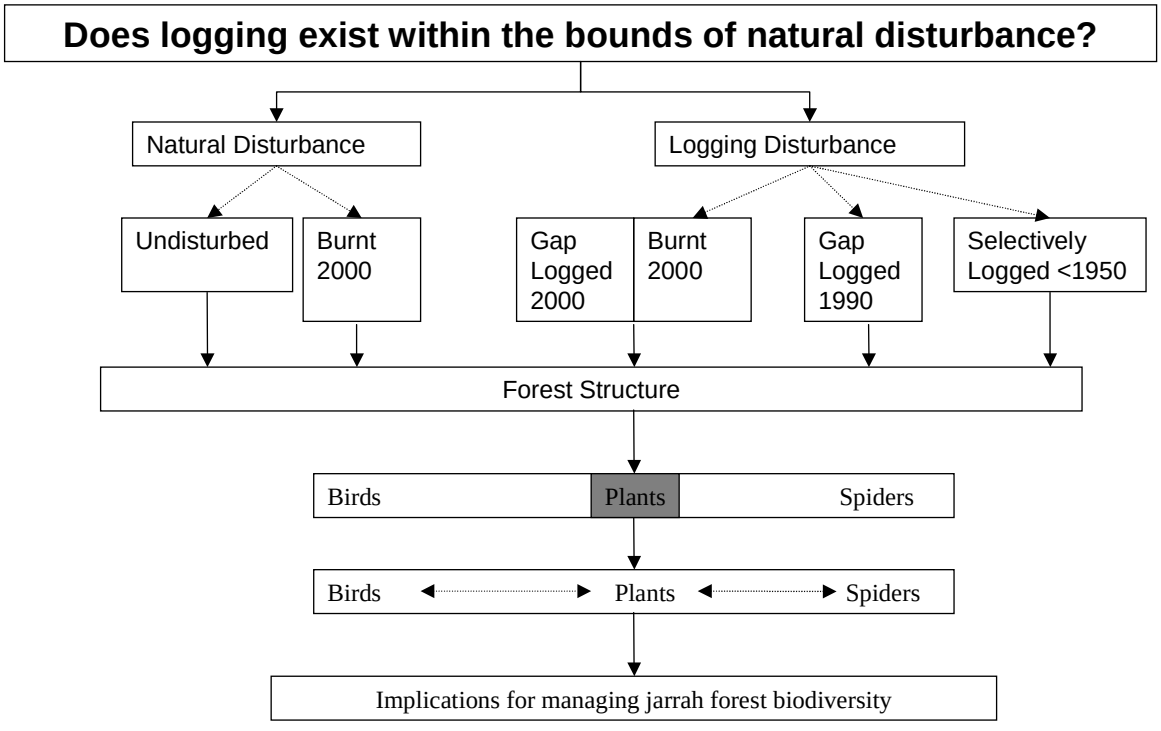
Forest selectively logged prior to 1950 were structurally very different to forest that has been disturbed by gap logging in 1990 and 2000. Forest selectively logged prior to 1950 has more jarrah dominant trees, jarrah decay 4 trees, subdominant banksia cover, marri dominant cover, decay class 4 logs and a greater volume of wood in stumps than forest disturbed by gap logging. In contrast, forest disturbed by gap logging has a much greater volume of coarse woody debris which is predominantly medium and large jarrah logs of decay class 2, and, to a lesser degree, decay class 3 jarrah logs and balga logs. Although the absence of 50 year-old gap logged forest prevents direct comparison, these differences in tree and CWD attributes show that gap logging is a more intensive form of logging than previous selective regimes.

4.4.4 Conclusion

Logging disturbance affects different structural attributes than fire disturbance. This research showed that autumn fire at Mt Dale destroyed flammable ground attributes but did not influence the size and decay structure of trees and the volume of CWD. Other studies in jarrah forest have shown these attributes recover 7 years after fire. In contrast, logging influences the size and decay structure of trees, and the volume of CWD, for at least 50 years after selective logging. The much greater increase in CWD and decrease in stand basal area of trees in forest gap logged in 1990 compared to forest selectively logged prior to 1950 suggests that contemporary gap logging creates greater structural change than past selective logging practices. The slow growth rate of jarrah trees and decay of jarrah logs leads to the prediction that structural differences will occur for centuries following gap logging disturbance.

CHAPTER 5

Understorey Plants



Chapter 5

5.1 Introduction

Understorey plants were selected as one of four surrogates of forest biodiversity in this study. They were selected for several reasons as outlined in Section 3.6.2. Studies in other temperate forests show that the species richness and composition of understorey plant communities can be affected by logging or fire disturbance (e.g., Hickey 1994; Halpern & Spies 1992; Decocq *et al.* 2004). Changes to canopy cover, volume of coarse woody debris, soil properties and leaf litter complexity that result from different disturbances has been shown to affect understorey plant diversity in many temperate forests (see Fig. 3.5 p. 55). In the forests of southwest Australia, the understorey is extremely species rich (estimated at least 2500 species by N.G. Marchant in Gioia & Pigott 2000). Almost 80 % of these species are endemic to Western Australia (Department of Conservation and Land Management 2004). Natural and anthropogenic disturbances have been shown to be important factors that determine the distribution and abundance of these species (Havel 1979; Burrows & Friend 1998; Burrows *et al.* 2002b; Yates *et al.* 2003). The high level of species richness, and the responsive nature of understorey communities to different disturbance regimes, makes them a useful study focus for forest ecology. Therefore, understorey plants were chosen to investigate the effect of fire and logging disturbance at Mt Dale.

Many studies have investigated how fire affects understorey plant communities in jarrah forest (see reviews by Christensen & Abbott 1989; Bell *et al.* 1989; Burrows 2002). These studies have highlighted that the vast majority of species contain characteristics indicative of a long term association with regular disturbance by fire. These characteristics include subterranean organs that

regenerate after fire and seeds that are stimulated to germinate after fire (Christensen & Kimber 1975; Bell & Koch 1980; Dixon *et al.* 1995; Roche *et al.* 1997; Ward *et al.* 1997; Burrows & Wardell-Johnson 2003). These characteristics have allowed species to survive in an environment that has been regularly burnt over the last 50 millennia (Pattinson *et al.* 1999; Section 2.2). In fact, seedling germination and successful establishment is very rare in the absence of fire or other disturbances in jarrah forest (Burrows & Friend 1998).

Plant species are adapted to a particular fire regime, incorporating fire frequency, intensity and season (Gill 1975; Bond & van Wilgen 1996). In jarrah forest, fire frequency (e.g., Christensen & Abbott 1989; Burrows & Wardell-Johnson 2003), fire intensity (e.g., Peet 1971; Christensen & Kimber 1975; Shea *et al.* 1979; Smith *et al.* 2000) and season (e.g., McCaw 1988; Grant & Loneragan 1999b) affect the species richness and composition of understorey species. With respect to species richness, the highest number of species is hypothesised to occur when a site simultaneously contains species stimulated to germinate by fire and species that resprout following fire (Bell *et al.* 1987). In a retrospective study that assessed plant diversity across 27 sites representing a continuum from four months to 46 years since the last fire, the highest number of species occurred between 2 and 5 years after fire (Bell & Koch 1980). More recently, Burrows & Wardell-Johnson (2003) showed that species richness peaked after 2 years in intermediate rainfall (1000 mm yr^{-1}) and after 7 years in low rainfall areas (750 mm yr^{-1}). Burrows & Wardell-Johnson (2003) also showed that long-term monitoring (> 30 years) of jarrah forest has shown that no fire regime is optimal for all plant species and that species richness and composition vary with time since fire and fire regime. The importance of the relationship between fire

regimes and the assemblage of species within a community is complex, and further research is required to improve the understanding of the significance of fire to management of understorey diversity (Yates *et al.* 2003).

In contrast to fire, where many studies have been undertaken, there has been only one published study that has assessed the affect of logging disturbance on understorey plant communities. The Kingston study (described in Section 2.5) found a similar species richness of native plants at the coupe scale when comparing unlogged forest to adjacent patches that were logged four years previously (Burrows *et al.* 2002b). In contrast, in small plots ($1m^2$) the average species richness is 20 - 30% higher in unlogged buffers compared with that in recently logged forest (Burrows *et al.* 2002b). Furthermore, the abundance (number of individual plants) of native plants was higher in unlogged buffers, whereas the abundance of weeds was higher in logged patches. As the Kingston study only assessed the short-term impact of logging (up to four years post logging), it is not known how long it takes plant communities to recover from this disturbance.

Understorey plant species differ in susceptibility to disturbance, depending on particular life-history traits and strategies, such as regeneration, growth and dispersal (Decocq *et al.* 2004). Guilds that differentiate between species based on their response to fire have been successfully used to investigate the effects of logging in jarrah forest (Burrows *et al.* 2002b) and other fire prone temperate forests (e.g., Loyn *et al.* 1983; Murphy & Ough 1997). These studies show that different guilds respond uniquely to logging disturbance, and that significant effects may have been overlooked without the use of such guilds. The use of fire

guilds is particularly relevant in investigations that assess how jarrah forest plant communities are affected by logging because logging changes fire intensity as a result of increases in fine and coarse woody debris (Hobbs 2003; also see Chapter 4). The importance of fire to the reproduction and regeneration of jarrah forest understorey species means that changes in fire intensity are likely to influence the composition of species that exist after logging disturbance.

This chapter will address the aims of this thesis outlined in Section 2.7 by assessing three questions:

1) Are there differences in the richness or assemblage of understorey plant species within different guilds in forest recently disturbed by fire compared with forest recently disturbed by gap logging and fire?

2) Does the richness or assemblage of understorey plant species within different guilds recover 10 years after gap logging?

3) Does the richness or assemblage of understorey plant species within different guilds recover 50 years after selective logging?

5.2 Methods

5.2.1 Selecting Guilds

As outlined in Section 3.6.2, plants were placed within guilds. Plant species of the jarrah forest can be differentiated into two groups based on their response to fire (Bell *et al.* 1993; Burrows & Wardell-Johnson 2003):

1/ Obligate seeding species. The reproductively mature plants of this group die following stem girdling, or 100% leaf scorch, by fire with populations re-establishing from seed (Burrows & Wardell-Johnson 2003). Approximately one-third of the species in the jarrah forest are obligate seeders (Burrows & Wardell-Johnson 2003). Fire frequency is an important determinant of the species richness of obligate seeders in jarrah forest. If fires are too frequent, seeding species can be eliminated because they do not have sufficient time to replenish their soil seed bank (Grant & Loneragan 1999b). When considering logging impacts, factors such as disturbance to soil, post-logging burning, and increased populations of seed harvesting animals (e.g., ants) are likely to impact on obligate seeding species (de Grandpre *et al.* 2000; Smith *et al.* 2000; Burrows *et al.* 2002b).

2/ Resprouting species. The reproductively mature plants of these species survive fire and there is usually limited investment into reproductive output by seeds (Burrows & Wardell-Johnson 2003). These plants survive fire by resprouting from vegetative buds that are protected under the bark of above-ground organs, lignotubers, or underground perenniating tissue. Approximately two thirds of the understorey species in jarrah forest are resprouting species (Burrows & Wardell-Johnson

2003). In contrast to obligate seeding plants, resprouting species are favoured by reasonably frequent burning but may decline in areas that remain unburnt for a long period of time (Bell *et al.* 1989). Mining in jarrah forest areas has been shown to have a deleterious impact on regenerating species because there is complete disturbance of all former vegetation so that resprouting is largely eliminated (Bellairs & Bell 1993; Koch & Ward 1994). The impact on the soil of mechanical disturbance and the possible drying out of the soil following logging are suspected as having an impact on regenerating species in jarrah forest (Burrows *et al.* 2002b), and have been shown to affect regenerators in Mountain Ash (*Eucalyptus regnans*) forest (Murphy & Ough 1997).

Logging disturbance influences soil properties in addition to influencing fire properties (Hingston 1985; Incerti *et al.* 1987; Hunter 1999). Burrows *et al.* (2002b) suggested that the heavy machinery used in contemporary logging practices in jarrah forest may influence between 60 – 80 % of the area in logged coupes. As disturbance to soil properties will impact on the species richness and composition of obligate seeders and regenerators (Burrows *et al.* 2002b), species within each guild were further differentiated into different categories based on where a plant regenerated in unfavourable conditions (e.g., above or below the soil) (Fig. 5.1). Species were classified using the life form classes of Pate and Dixon (1982) who modified the widely acknowledged system of Raunkaier (1934). Species were placed into one of four major life forms:

1) **Nanophanerophytes:** have perenniating buds on upright shoots above the ground level.

2) **Hemicryptophytes:** perennial, usually herbaceous, plants that bear buds at ground level.

3) **Cryptophytes:** herbaceous perennials with buds below soil level. Cyptophytes are regenerating plants.

4) **Therophytes:** ephemeral or annual plants that flower and die at the end of a short growing season. Therophytes are obligate seeding plants.

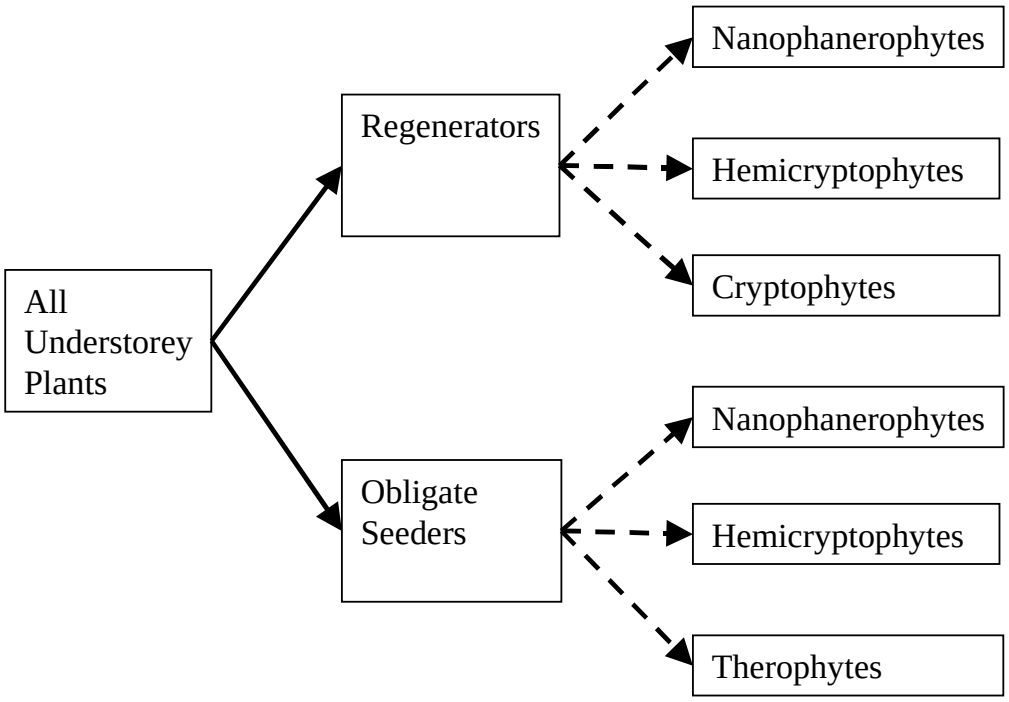


Fig. 5.1 Flow diagram showing how different plant species were categorised in guilds.

5.2.2 Site Design and Specimen Collection

Pilot studies conducted in March 2001 showed that the understorey plant community contained between 80 and 150 species within each one hectare site. A similar range in species richness was also noted by Havel (1979). This pilot study showed that it was difficult and time consuming to sample all species within the 1 ha area. The Integrated Forest Monitoring System (I.F.M.S.) Concept Plan (1999) showed that sub-sampling sites using smaller plots was an appropriate way to assess whether changes in plant diversity could be attributed to different disturbance regimes (CALM 1999). At each site, four 400 m^2 (20m x 20 m) plots were used to sample plant diversity. Plots were placed at a random distance between the centre of the site and each of the four corners, and were marked with wooden stakes and flagged (Fig. 5.2). These four plots were combined to sample the plant species at each site. Therefore, plant diversity was measured within 0.16 ha of each site.

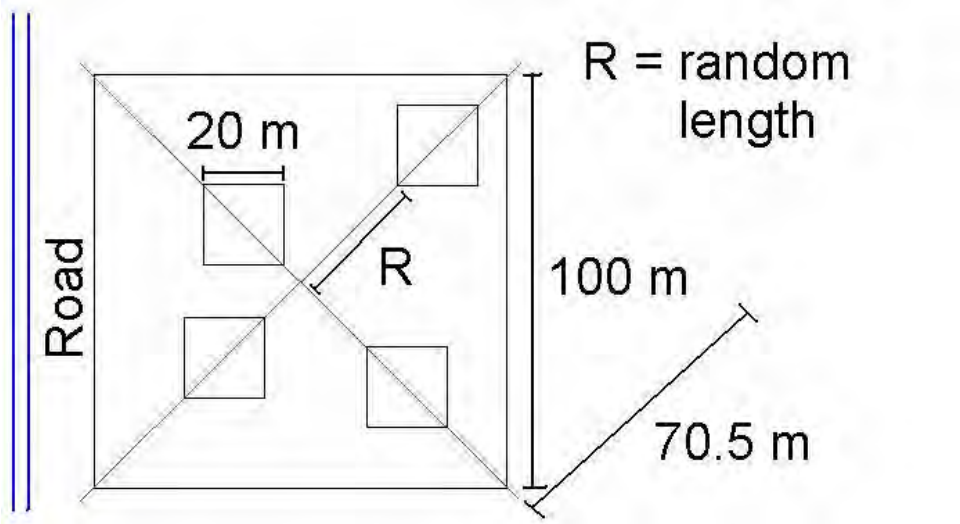


Fig. 5.2 Sampling design used for the flora in each treatment site. Each site was one hectare in size and contained four quadrates, each 20 x 20 metres in size. Quadrats were placed at random distances between the centre and the corners of each site.

Within each plot, representatives of all vascular understorey plant species (< 2m in height) were collected and recorded. Where possible, three specimens were taken of each species found (one voucher specimen for WA Herbarium, one Authority specimen and one specimen for identification). Plant specimens were collected between early September and early November 2000, corresponding with the main flowering period for this Botanical District. Species were identified using Blackall and Grieve (1974, 1980, 1981, 1985), Grieve (1982, 1998), Grieve and Blackall (1982), Marchant *et al.* (1987), Marshall (1990) and Meney and Pate (1999). Specimens were verified using the plant reference collection at the CALM Western Australian Reference Herbarium at Kensington. Authorities for every species are given in the species in Appendix 3.

5.2.3 Statistical Approach

As outlined in Section 3.6.2, two measures of diversity were used to assess whether there were differences between natural and logging disturbances (α diversity - species richness within sites, and β diversity - compositional changes between sites). The 'presence' or 'absence' of species at each site was used to calculate each measure of species diversity. Although presence/absence analysis does not have the resolution to capture all potentially important changes in populations, it was selected because managers operate within the constraints that no species should be lost (become extinct) from any area (Wildlife Conservation Act 1950; Conservation and Land Management Act 2000). This research assessed the possibility of loss of species after logging by examining whether the number of species declines after logging (by assessing species richness) and examining whether changes to species composition could be attributed to past logging practices (species composition).

1. Species Richness

Species richness data were analysed using univariate statistics. Initially, trends of the average number of species for each treatment were described using column graphs. Analyses of Variance (ANOVAs) were then performed on this data using the SPSS statistical package (version 10). The data was transformed using natural log because it was count data (Fowler & Cohen 1990). The three major assumptions of ANOVA addressed in Section 4.2.3 were also addressed in this section. Where significant differences were found in the ANOVA, *post hoc* analysis comparing between means using Games-Howell statistics was conducted on the SPSS package as the variances were not assumed to be equal.

A similar cautious approach was taken with regards to interpreting each ANOVA in this chapter as was outlined in Section 4.2.3. In my research, the null hypothesis (H_o) and alternative hypothesis (H_a) for each guild is:

H_o : There is no difference in species richness among the five treatments.

H_a : There are differences in species richness among the five treatments.

Three steps were taken for each ANOVA to ensure that the chance of making a Type II error was reduced:

1) When the probability value was less than 5 % (i.e., $p < 0.05$), then alternate hypothesis was accepted.

2) When the probability value was between 5 and 10% ($0.05 < p < 0.1$), the statistical power of the ANOVA was scrutinised. If the test did not have the statistical power to reject the null hypothesis if it was truly false (i.e., power < 0.8 *sensu* Strehlow *et al.* 2002) than the probability value for the ANOVA was increased from 0.05 to 0.l. This change decreases the risk of making a Type 11

error but increases the risk of making a Type 1 error (i.e., accepting Ha when it is truly false) (Calver 2003).

3) When statistical power was not sufficient (power < 0.8), the number of sites was doubled, quadrupled (etc) until statistical power was satisfactory (power > 0.8). This design was used to show how many sites are required to reduce the possibility of committing Type II error (pers. comm. Calver 2003).

2. Species Composition

The composition of each guild was assessed using multivariate techniques on the PRIMER statistical package. An analysis of similarity (ANOSIM) using Bray-Curtis dissimilarity measures (Czehanowski coefficient) was performed to test whether there were differences in community composition between each treatment. There are no assumptions that are req uired to be met for ANOSIM analysis (Clarke & Warwick 1994). Where differences were found, SIMPER analysis was performed to assess which species were driving the observed dissimilarity between treatments. The output comprised a list of species in decreasing order of importance which discriminates between each of the two statistically significant treatments. Species that are listed in each SIMPER table are those that contributed greater than 5% of the variation as determined by SIMPER analysis. Ordinations were also conducted using non-metric multi-dimensional scaling (nMDS) (Kruskal and Wish 1978). The number of dimensions used was the minimum required to achieve a stress < 0.2 (as recommended by Clarke & Warwick 1994).

5.3 Results

5.3.1 General Description

Two hundred and twenty-six understorey taxa representing 44 families were recorded across all treatments. One hundred and thirty-nine species regenerate following fire. Of these regenerating species, 62 were nanophanerophytes, 17 were hemicryptophytes and 58 were cryptophytes. Less than a quarter of the 226 taxa were identified as obligate seeding species (n = 43), of which 28 were nanophanerophytes, 12 were hemicryptophytes and three were therophytes. These three therophytes were not investigated further because of low sample size. The remaining 44 taxa were not classified because they could not be identified to a species level (n = 25) or because of a lack of biological information (n = 19) (Appendix 3). No species were exotic.

Most species were uncommon and patchy in distribution. Approximately one-third of the total number of species were recorded at each site ($\bar{x}$ = 58.7 SE ± 2.5 species/site, range = 49 - 84 species/site). Only five species of the 182 species were recorded at all sites (representing < 3 % of the species assessed in this research). These species were *Dryandra lindleyana, Eucalyptus marginata, Bossiaea ornata, Hovea chorizemifolia* and *Xanthorrhoea gracilis.* In contrast, 102 species (56 % of the species assessed in this research) were recorded at fewer than four of the 20 sites. Of these 102 species, 43 % were recorded in undisturbed forest, 34 % in forest burnt in 2000, 31 % in forest selectively logged forest < 1950, 22 % in forest gap logged and burnt in 2000 and 15 % in forest gap logged in 1990.

There were differences in the average number of species per site among the five treatments ($p < 0.05$; Table 5.1). Pairwise analysis showed that more species were recorded in undisturbed forest than forest that were gap logged in 1990, gap logged and burnt in 2000, and forest that was burnt in 2000 ($p < 0.05$; Fig. 5.3). In contrast, there were no differences in the average number of species between sites located in undisturbed forest and those in forest selectively logged prior to 1950 ($p > 0.05$; Fig. 5.3). There were also no differences between the number of species in forest that was gap logged in 1990, forest that was gap logged/ burnt in 2000 and forest burnt in 2000 ($p > 0.05$; Fig. 5.3).

Table 5.1 One-way ANOVA and power analysis testing whether there are differences in the species richness understorey plants between treatments representing different disturbance regimes.

	Sum of Squares	Degrees of Freedom	Mean Square	F-ratio	P Value	POWER
Between Groups	.137	4	.034	4.567	0.01	0.71
Within Groups	.113	15	.008			
Total	.250	19				

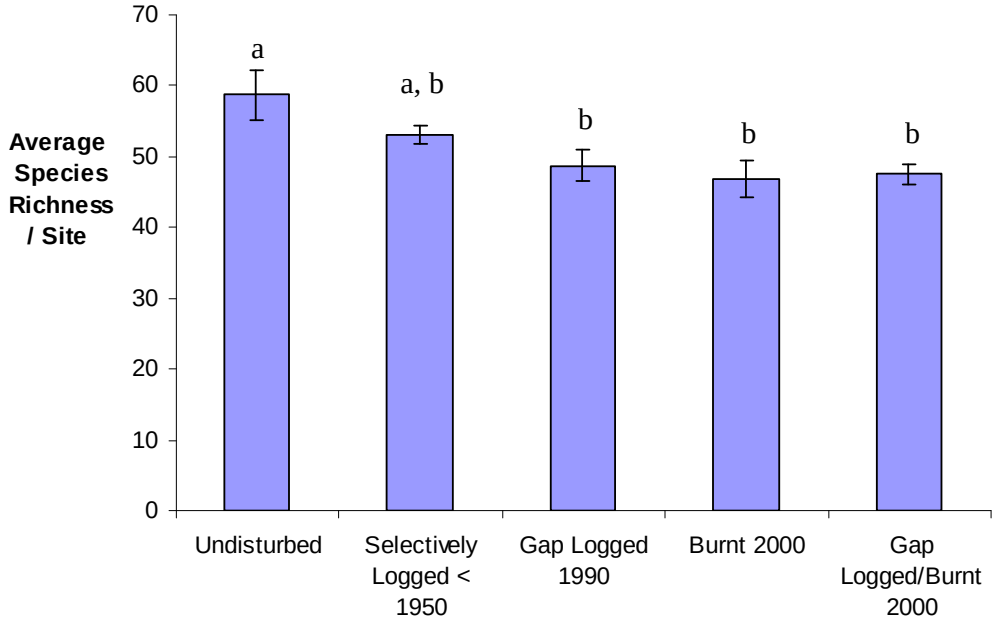


Fig. 5.3 The average (± SE) species richness of understorey plants for five treatments representing different disturbance regimes. Significant differences between sites were determined by oneway ANOVA. Different letters indicate significant differences ($p < 0.05$).

Multivariate analysis showed that there were distinct differences in the composition understorey species among treatments (Global $R = 0.27$, $p < 0.05$).

Pairwise analysis shows that both treatments burnt in 2000 ('burnt 2000' and 'gap logged/burnt 2000') had a different composition of plants than undisturbed forest and forest selectively logged prior to 1950 (Table 5.2). Furthermore, the composition of species in forest that was gap logged in 1990 was different to communities found in forest gap logged and burnt in 2000, but not to the communities found in forest that was burnt in 2000 (Table 5.2).

Table 5.2 Summary of ANOSIM pairwise analysis testing whether the composition of understorey plant communities was different between five treatments representing different disturbance regimes. Highlighted rows denote significant differences between treatments ($p < 0.05$).

Comparison between Treatments	Global *R*	P Value
Undisturbed, Burnt 2000	0.48	0.029
Undisturbed, Gap Logged/ Burnt 2000	0.33	0.029
Undisturbed, Gap Logged 1990	-0.01	0.543
Undisturbed, Selectively Logged <1950	0.09	0.771
Burnt 2000, Gap Logged/ Burnt 2000	0.385	0.057
Burnt 2000, Gap Logged 1990	0.563	0.057
Burnt 2000, Selectively Logged <1950	0.688	0.029
Gap Logged/ Burnt 2000, Gap Logged 1990	0.469	0.029
Gap Logged/ Burnt 2000, Selectively Logged <1950	0.365	0.029
Gap Logged 1990, Selectively Logged <1950	-0.083	0.629

Ordination depicts the differences between communities found in forest that were burnt in 2000 ('burnt 2000' and 'gap logged/burnt 2000' treatments) and sites that had not been burnt for five years ('undisturbed', 'selectively logged < 1950' and 'gap logged 1990') (Fig. 5.5). There also appeared to be different communities in forest gap logged and burnt in 2000 compared to forest burnt in 2000 (Fig. 5.5).

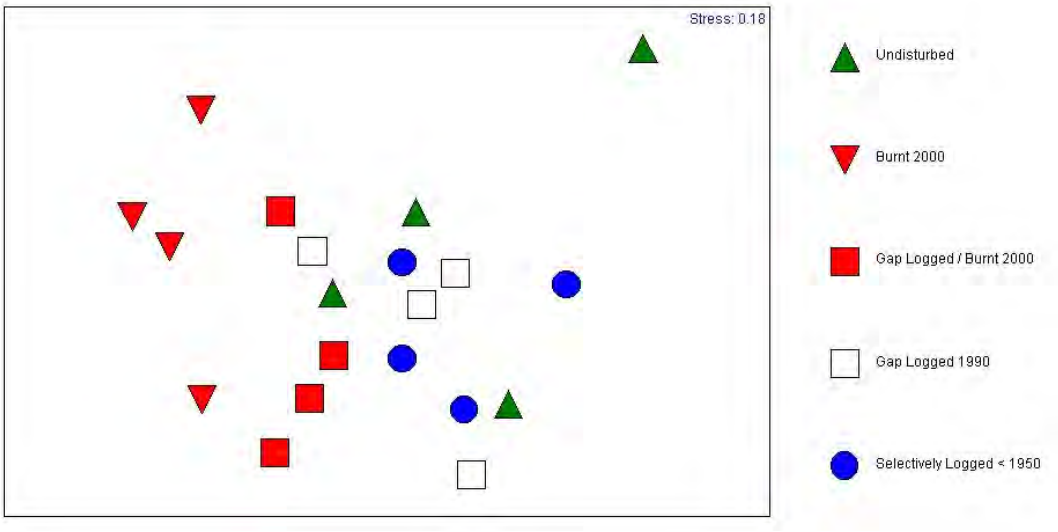


Fig. 5.4 Non-metric multidimensional scaling plot showing the effect of five disturbance regimes on the composition of all understorey species.

When sites are pooled within forest of similar disturbance regimes, undisturbed forest contained more species than all other treatments (Fig. 5.5). Forest selectively logged prior to 1950 contained more species that forest gap logged in 1990 (Fig. 5.5). With respect to the two treatments burnt in 2000, unlogged forest contained more species than forest that was gap logged (Fig. 5.5). The unlogged forest burnt in 2000 ('Burnt 2000') had more species than forest gap logged in 1990 (Fig. 5.5).

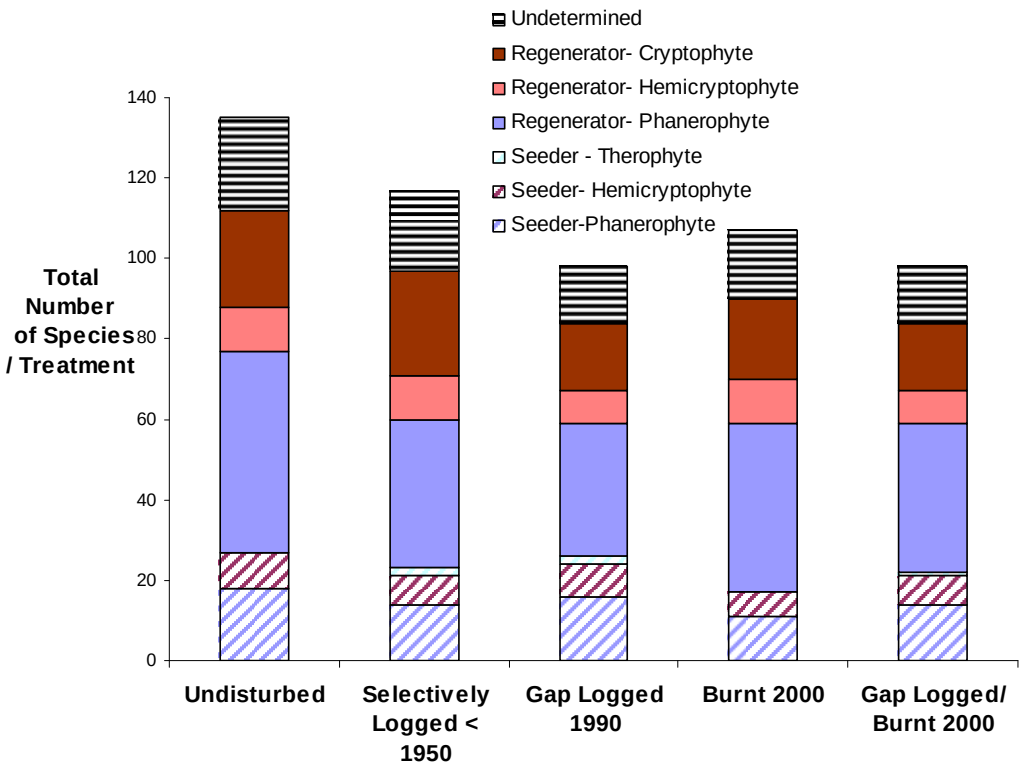


Fig. 5.5 The total number of understorey plant species recorded within each treatment. Each treatment consisted of four sites. Plants are grouped within guilds.

5.3.2 Obligate Seeders – Nanophanerophytes

Species Richness

The number of seeding nanophanerophytic species varied between sites (range 4 - 11, $\bar{x}$ = 7.1 SE ± 0.48, species per site). No statistical difference in species richness was found among all treatments at standard levels of significant ($p < 0.05$; Table 5.3). *Post hoc* analysis revealed that the experiment contained low power (power = 0.55). Eight sites within each treatment are required to detect a large effect size using standard levels of significance (i.e., power > 0.8). When accounting for the low statistical power, there were differences among treatments ($p < 0.1$) (Table 5.3). Pairwise analysis showed that sites in undisturbed forest had on average more species than sites in treatments disturbed by fire in 2000 ('burnt 2000' and 'gap logged/burnt 2000') ($p < 0.1$; Fig. 5.6). There was no

difference in the number obligate seeding nanophanerophytes in undisturbed forest, forest selectively logged prior to 1950 and forest gap logged in 1990 ($p > 0.1$; Fig. 5.6). There was also difference in the number of obligate seeding nanophanerophytes between the two treatments burnt in 2000 ('gap logged/burnt 2000' and 'burnt 2000') and forest selectively logged prior to 1950 and forest gap logged in 1990 ($p > 0.1$; Fig. 5.6).

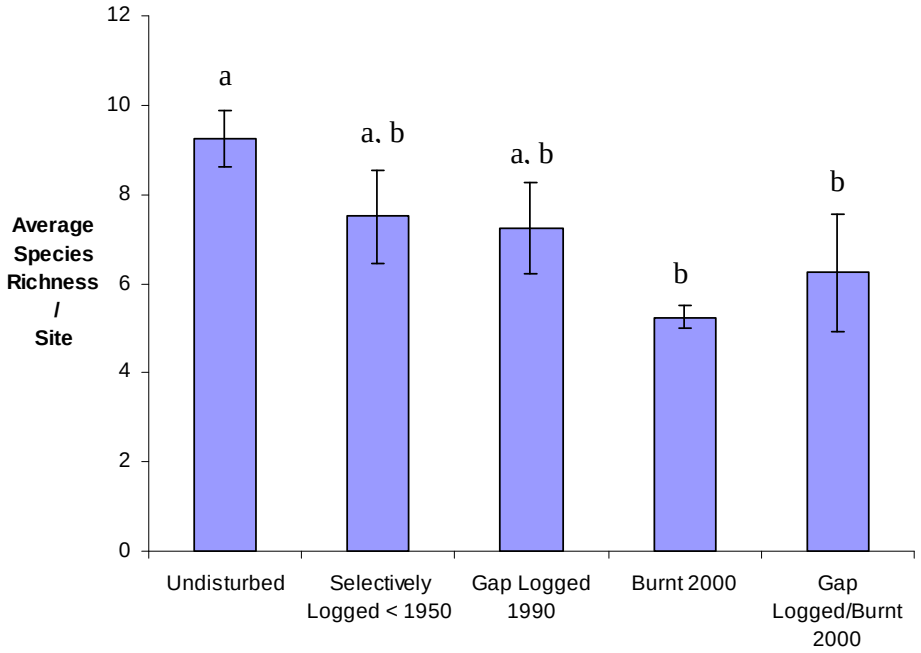


Fig. 5.6 The average (± SE) species richness of obligate seeding nanophanerophytes per site for five treatments representing different disturbance regimes. Significant differences between sites were determined by one-way ANOVA. Different letters indicate significant differences ($p < 0.1$).

Table 5.3 One-way ANOVA and power analysis testing whether there are differences in the species richness of obligate seeding nanophanerophytes between treatments representing different disturbance regimes.

	Sum of Squares	Degrees of Freedom	Mean Square	F-ratio	P Value	POWER
Between Groups	0.76	4	0.19	2.43	0.09	0.55
Within Groups	1.12	15	0.08			
Total	1.92	19				

Species Composition

There were significant differences in the composition of obligate seeding nanophanerophytes between the five treatments (Global $R = 0.23$, $p < 0.05$). Pairwise analysis showed that the two treatments disturbed by fire in 2000

('burnt 2000' and 'gap logged/burnt 2000') contained a different community of regenerating nanophanerophytes than found in undisturbed forest (Table 5.4). There no differences in composition among any of the other treatments (Table 5.4). Some trends may have been masked by the large variation in species composition between sites. Sixteen species (57% of obligate seeding nanophanerophytes) were recorded in less than 4 sites, creating considerable variation in species composition within treatments (Table 5.5; Fig. 5.7). Undisturbed forest contained the highest number of these uncommonly recorded species (n = 8) followed by gap logged 1990 (n = 7), gap logged/ burnt 2000 (n = 5) and selectively logged prior to 1950 (n = 4). Burnt forest had the lowest number of uncommonly recorded species (n = 2). Sites burnt in 2000 were different to sites located in undisturbed forest on the ordination (Fig. 5.7). Sites that were in forest selectively logged prior to 1950 and gap logged in 1990 appear as a statistical boundary between undisturbed sites and sites burnt in 2000 (Fig. 5.7).

Table 5.4 Summary of ANOSIM pairwise analysis testing whether the composition of obligate seeding nanophanerophytes were different between five treatments representing different disturbance regimes. Highlighted rows denote significant differences between treatments ($p < 0.05$).

Comparison between Treatments	Global R	P Value
Undisturbed, Burnt 2000	0.69	0.029
Undisturbed, Gap Logged/ Burnt 2000	0.41	0.029
Undisturbed, Gap Logged 1990	-0.09	0.686
Undisturbed, Selectively Logged <1950	0.07	0.400
Burnt 2000, Gap Logged/ Burnt 2000	0.02	0.457
Burnt 2000, Gap Logged 1990	0.42	0.086
Burnt 2000, Selectively Logged <1950	0.33	0.057
Gap Logged/ Burnt 2000, Gap Logged 1990	0.07	0.257
Gap Logged/ Burnt 2000, Selectively Logged <1950	0.23	0.143
Gap Logged 1990, Selectively Logged <1950	0.16	0.314

Table 5.5 The obligate seeding phanerophyte recorded in five treatments representing different disturbance regimes ('Un' – Undisturbed, 'SL 50' – Selectively Logged < 1950, 'GL 90' – Gap Logged 1990, 'GLB' – Gap Logged/ Burnt 2000, 'B' – Burnt 2000). Numbers represent the number of sites that species were recorded.

FAMILY	SPECIES	Un	SL 50	Gl 90	GLB	B
AMARANTHACEAE	*Ptilotus esquamatus*	0	0	0	0	2
	Ptilotus polystachyus	2	0	0	0	0
ANTHERICACEAE	*Laxmannia squarrosa*	2	0	1	0	0
	Laxmannia sp. 1	1	1	0	1	0
EPACRIDACEAE	*Styphelia tenuiflora*	3	4	4	4	4
FABACEAE	*Gastrolobium calycinum*	0	1	0	0	0
	Gompholobium knightianum	3	4	3	3	1
	Gompholobium marginatum	1	3	0	1	3
	Gompholobium preissii	2	0	1	2	1
	Kennedia coccinea	3	2	3	3	0
MIMOSACEAE	*Acacia alata*	1	0	0	0	0
	Acacia applanata	1	0	2	0	0
	Acacia lasiocarpa	0	0	0	2	3
	Acacia pulchella	2	1	1	1	1
	Acacia stenoptera	0	0	0	1	0
	Acacia urophylla	0	2	1	0	0
	Acacia sp.1	1	0	1	0	0
	Acacia sp. 2	0	0	0	1	0
PROTEACEAE	*Dryandra sessilis*	4	2	2	0	0
	Hakea undulata	1	1	0	0	0
RHAMNACEAE	*Trymalium floribundum*	0	0	1	0	0
	Trymalium ledifolium	2	2	0	0	0
SANTALACEAE	*Leptomeria cunninghamii*	3	2	3	0	1
STACKHOUSIACEAE	*Tripterococcus brunonis*	1	0	1	0	1
THYMELEACEAE	*Pimelea spectabilis*	0	1	1	2	2
	Pimelea suaveolens	0	0	1	1	0
TREMANDRACEAE	*Tetratheca hirsuta*	4	4	3	2	2
VIOLACEAE	*Hybanthus floribundus*	0	0	0	1	0
Total Number of Species / Treatment		**18**	**14**	**16**	**14**	**11**

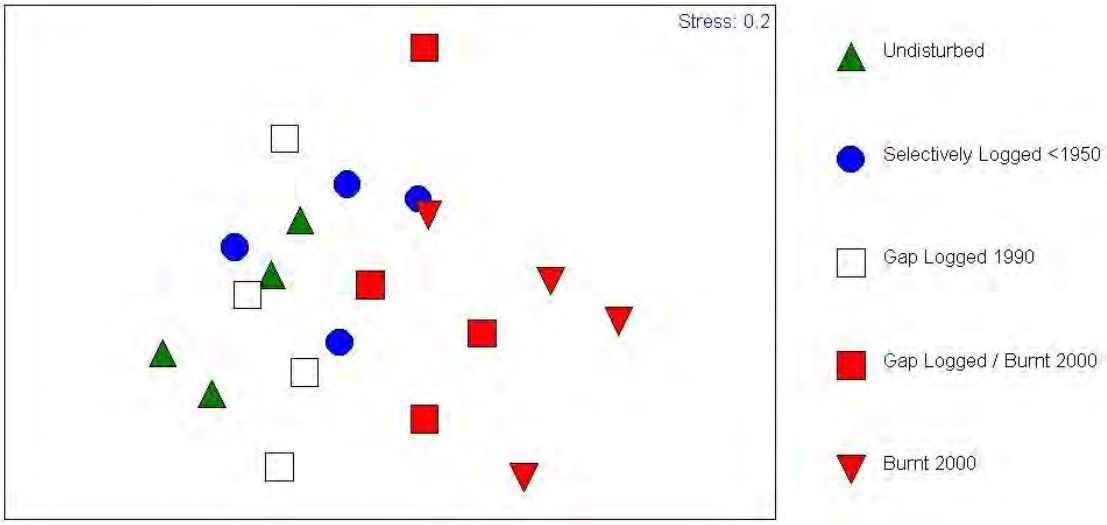


Fig. 5.7 Non-metric multidimensional scaling plot showing the effect of five disturbance regimes on the composition of obligate seeding nanophanerophytes.

SIMPER analysis showed that of the 28 obligate seeding nanophanerophytic species, six species contributed to differences in species composition between undisturbed forest and forest that had been recently disturbed by fire. *Dryandra sessilis* did not occur in fire affected sites, and *Leptomeria cunninghammi* and *Tetratheca hirsuta* were recorded less frequently in these sites (Table 5.6). In contrast, *Acacia lasiocarpa* was found more often in forest that was burnt in 2000 than in forest that had not been burnt for five years (Table 5.5; Table 5.6). Two species had different distributions when comparing between forest that was burnt in 2000 with forest that was logged and burnt in 2000. *Kennedia coccinea* and *Gompholobium knightianum* were recorded more frequently in forest gap logged and burnt in 2000 that forest burnt in 2000 (Table 5.5; Table 5.6).

Table 5.6 The obligate seeding nanophanerophytes that contributed to differences between paired treatments identified as containing different assemblages of species by ANOSIM analysis (Table 5.4). Species are grouped within the treatment they were more commonly encountered.

Treatments	**Species Contributing to Difference Between Treatments (% Contribution to Dissimilarity)**
Undisturbed	*Dryandra sessilis* (9.9) *Kennedia coccinea* (7.2) *Leptomeria cunninghammi* (6.1) *Gompholobium knightianum* (6.1) *Tetratheca hirsuta* (5.0)
Burnt 2000	*Acacia lasiocarpa* (7.4)
Undisturbed	*Dryandra sessilis* (10.5) *Leptomeria cunninghammi* (7.8) *Tetratheca hirsuta* (5.2)
Gap Logged / Burnt 2000	*Acacia lasiocarpa* (5.2)

5.3.3 Obligate Seeders - Hemicryptophytes

Species Richness

Low species richness and considerable variation in species number between sites characterised the seeding hemicryptophytic community (range 1 - 8, $\bar{x}$ = 4.2 SE ± 0.4, species per site) (Fig. 5.8). No statistical difference in species richness was found among all treatments at standard levels of significant ($p < 0.05$) (Table 5.7). However, the ANOVA had low statistical power (power = 0.56). Eight sites per treatment were required to detect a large effect size using standard levels of significance (i.e., power > 0.8). When the level of α was adjusted to account for this low statistical power, differences among treatments were apparent ($p < 0.1$) (Table 5.7). Pairwise analysis showed that forest that was burnt in 2000 had on average less species than all other treatments except forest that was gap logged in 1990 ($p < 0.1$; Fig. 5.8). The species richness of seeding hemicryptophytes was similar between sites located in forest gap logged in 1990, forest gap logged and burnt in 2000, forest selectively prior to 1950 and undisturbed forest was similar ($p > 0.1$; Fig. 5.8).

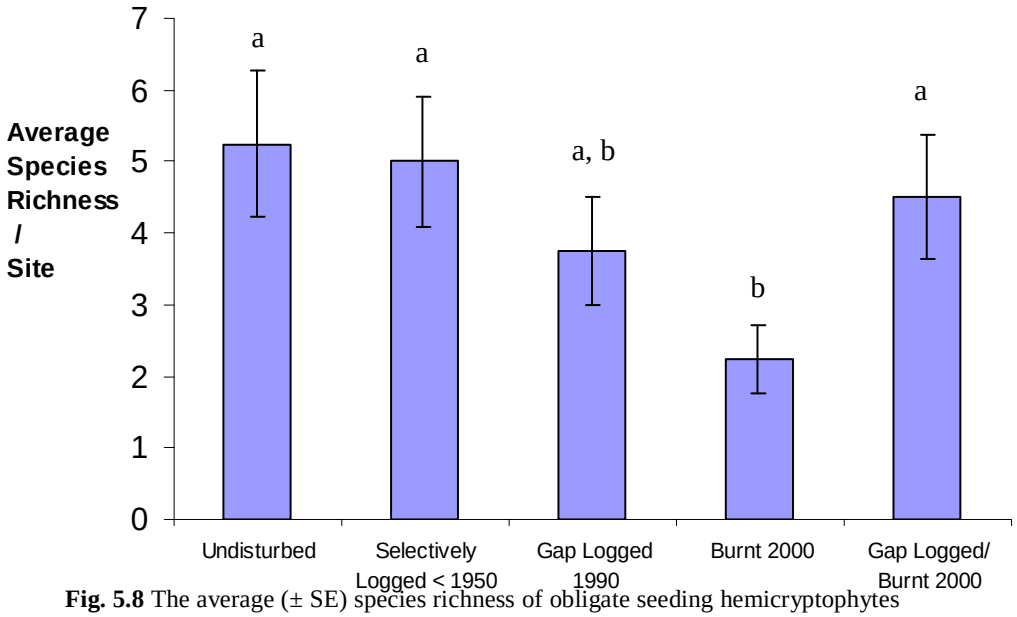


Fig. 5.8 The average (± SE) species richness of obligate seeding hemicryptophytes per site for five treatments representing different disturbance regimes. Significant differences between sites were determined by oneway ANOVA. Different letters indicate significant differences ($p < 0.1$).

Table 5.7 One-way ANOVA and power analysis testing whether there are differences in the species richness of obligate seeding hemicryptophytes between treatments representing different disturbance regimes.

	Sum of Squares	Degrees of Freedom	Mean Square	F-ratio	P Value	POWER
Between Groups	2.02	4	0.51	2.48	0.09	0.56
Within Groups	3.06	15	0.2			
Total	5.09	19				

Species Composition

The ANOSIM model found no difference in the composition of seeding hemicryptophytes among all treatments (Global $R = 0.05$; $p = 0.23$). There was no discernible trend among the five treatments on the ordination (Fig. 5.9). Obligate seeding hemicryptophytes species can be split into two groups. The first group contains the five species, *Ptilotus manglesii, Trichocline spathulata, Stylidium amoenum, Stylidium junceum* and *Stylidium schoenoides*, that together comprise over 80% of the records of obligate seeding hemicryptophytes (Table 5.8). All five species recorded in all treatments. The second group contains patchily distributed species. Five of these seven species were recorded in less

than 4 sites. These species were all from the genus *Stylidium*, and were distributed throughout all treatments (undisturbed n = 2 species (spp.), selectively logged prior to 1950 n = 1 spp., gap logged 1990 n = 2 spp., gap logged/burnt 200 n = 1 spp., burnt 2000 n = 1 spp.) (Table 5.8). Of the two remaining species, *Lagenofera huegelii* was found in undisturbed and both gap logged treatments, while *Neurachne alopecuroidea* was recorded at three undisturbed forest sites and two selectively logged sites, but not in any other treatments (Table 5.8).

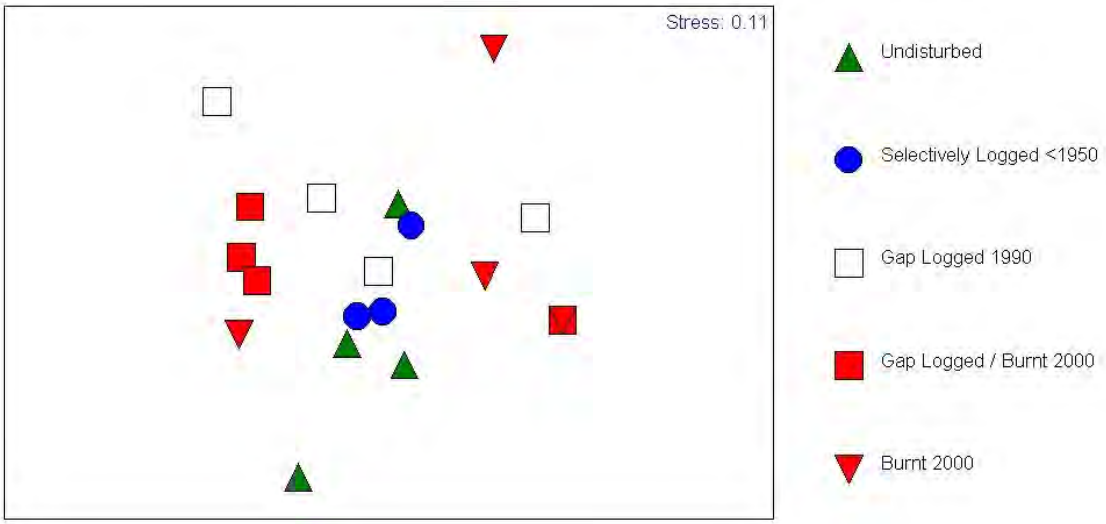


Fig. 5.9 Non-metric multidimensional scaling plot showing the effect of five disturbance regimes on the composition of obligate seeding hemicrytophytes.

Table 5.8 The obligate seeding hemicryptophytes recorded in five treatments representing different disturbance regimes ('Un' – Undisturbed, 'SL 50' – Selectively Logged < 1950, 'GL

90' – Gap Logged 1990, 'GLB' – Gap Logged/ Burnt 2000, 'B' – Burnt 2000). Numbers represent the number of sites that species were recorded.

FAMILY	SPECIES	Un	SL 50	Gl 90	GLB	B
AMARANTHACEAE	*Ptilotus manglesii*	2	4	3	3	1
ASTERACEAE	*Lagenofera huegelii*	1	0	1	2	0
	Trichocline spathulata	3	3	2	1	2
POACEAE	*Neurachne alopecuroidea*	3	2	0	0	0
STYLIDIACEAE	*Stylidium amoenum*	3	4	2	4	2
	Stylidium brunonianum	0	0	1	0	0
	Stylidium canaliculatum	1	0	0	0	0
	Stylidium ciliatum	0	0	0	3	0
	Stylidium diuroides	1	1	1	0	0
	Stylidium junceum	4	4	4	3	2
	Stylidium pycnostachyum	0	0	0	0	1
	Stylidium schoenoides	3	2	1	2	1
Total Number of Species / Treatment		9	7	8	7	6

Despite a lack of statistical difference, small changes in the composition of seeding hemicryptophytes were apparent between treatments that had experienced recent fire. There were fewer hemicryptophyte species in forest that was burnt in 2000 than in forest that was gap logged and burnt in 2000 (Fig. 5.8). *Stylidium ciliatum* was only recorded in gap logged and burnt forest (Table 5.8). Five other regularly recorded species (*Lagenofera huegelii, Ptilotus manglesii, Stylidium amoenum, Stylidium junceum* and *Stylidium schoenoides*) were recorded less frequently in burnt forest than gap logged/burnt forest (Table 5.8).

5.3.4 Regenerating Nanophanerophytes

Species Richness

There were on average 25 species of regenerating phanerophyte per site (SE ± 0.8, range = 22 – 37 species per site). Species richness was similar across all treatments (Fig. 5.10). One site (undisturbed site 2) had six more species than any other site, causing more variation in the undisturbed treatment than in the other four treatments (Fig. 5.10). No statistical difference in species richness was found among the five treatments at standard levels of significance ($p < 0.05$; Table 5.9). When the low statistical power in the ANOVA model ($p = 0.13$) was

taken into consideration, there was also no differences among treatments ($p > 0.1$) (Table 5.9). *Post hoc* analysis revealed that having eight times as many sites (i.e., 32 sites/ treatment) would be required to produce the required power in the model to detect a difference between treatments at standard levels of significance (power > 0.8 when n = 32 sites / treatment).

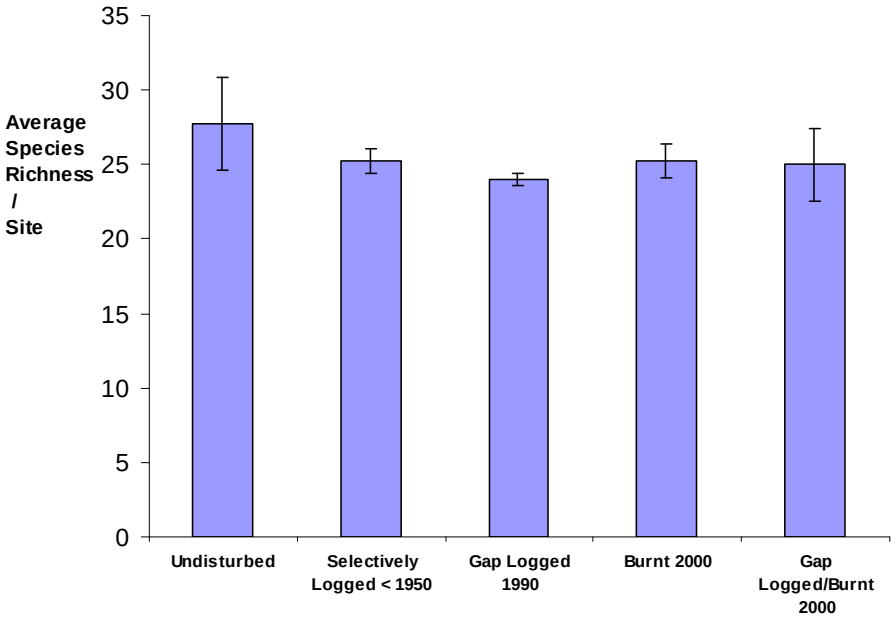


Fig. 5.10 The average (± SE) species richness of regenerating nanophanerophytes per site for five treatments representing different disturbance regimes. There were no significant differences between treatments ($p > 0.1$).

Table 5.9 One-way ANOVA and power analysis testing whether there are differences in the species richness of regenerating nanophanerophytes between treatments representing different disturbance regimes.

	Sum of Squares	Degrees of Freedom	Mean Square	F-ratio	P Value	POWER
Between Groups	0.04	4	0.01	0.47	0.76	0.13
Within Groups	0.30	15	0.02			
Total	0.33	19				

Species Composition

There were significant differences in the composition of regenerating nanophanerophytes between treatments (Global $R = 0.23$, $p < 0.05$) (Table 5.10). Both the ANOSIM and ordination show that forest burnt in 2000 ('burnt 2000' and 'gap logged/burnt 2000' treatments) contain a different assemblage of

species than forest gap logged in 1990 and forest selectively logged prior to 1950 (Table 5.10; Fig. 5.11a). Initially it was thought that the statistical similarity in the composition of undisturbed forest and the recently burnt treatments was due to the large variation caused by having an exceptionally high number of species found in undisturbed site 2 (Fig. 5.10). However, *post hoc* removal of this site from the analysis revealed no difference between undisturbed and recently burnt forest (Table 5.10b; Fig. 5.11 b).

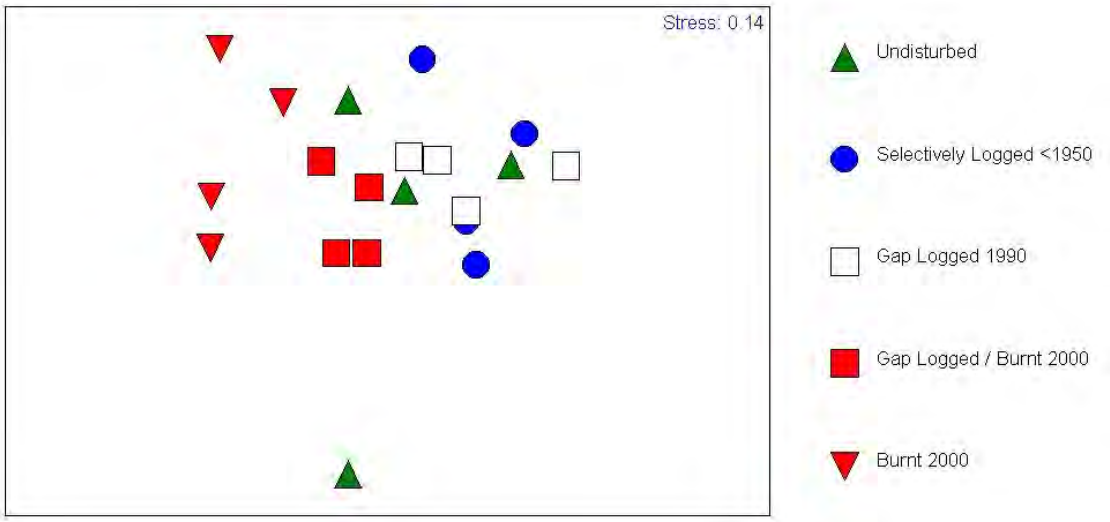
Table 5.10 Summary of ANOSIM pairwise analysis testing whether the composition of regenerating nanophanerophytes were different between five treatments representing different disturbance regimes. Table a) represents all sites, and Table b) represents all sites except U2, which had more species than other sites and was removed to analyse its influence. Highlighted rows denote significant differences between treatments ($p < 0.05$).

a) All Sites

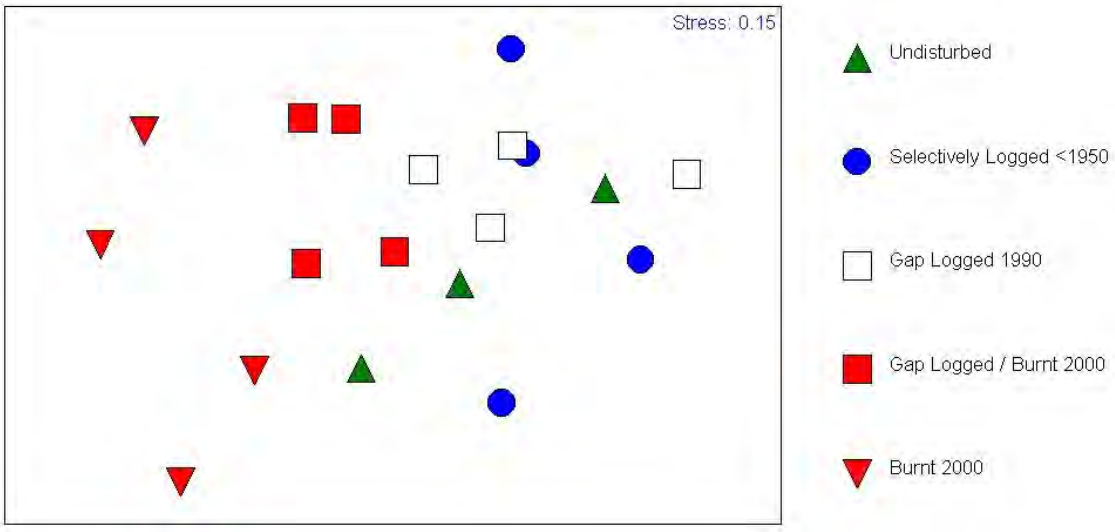
Groups	*Global* R	*Statistic %*
Undisturbed, Burnt 2000	0.22	8.6
Undisturbed, Gap Logged/ Burnt 2000	-0.01	48.6
Undisturbed, Gap Logged 1990	-0.07	71.4
Undisturbed, Selectively Logged <1950	-0.10	71.4
Burnt 2000, Gap Logged/ Burnt 2000	0.29	8.6
Burnt 2000, Gap Logged 1990	0.68	2.9
Burnt 2000, Selectively Logged <1950	0.87	2.9
Gap Logged/ Burnt 2000, Gap Logged 1990	0.49	2.9
Gap Logged/ Burnt 2000, Selectively Logged <1950	0.63	2.9
Gap Logged 1990, Selectively Logged <1950	-0.09	65.7

b) *Post hoc* removal of site 'U2'

Groups	*Global* R	*Statistic %*
Undisturbed, Burnt 2000	0.35	8.6
Undisturbed, Gap Logged/ Burnt 2000	0.09	34.3
Undisturbed, Gap Logged 1990	-0.10	62.9
Undisturbed, Selectively Logged <1950	-0.14	80.0
Burnt 2000, Gap Logged/ Burnt 2000	0.29	8.6
Burnt 2000, Gap Logged 1990	0.67	2.9
Burnt 2000, Selectively Logged <1950	0.87	2.9
Gap Logged/ Burnt 2000, Gap Logged 1990	0.49	2.9
Gap Logged/ Burnt 2000, Selectively Logged<1950	0.63	2.9
Gap Logged 1990, Selectively Logged <1950	-0.09	65.7



a) All Sites



b) Without site 'u2'

Fig. 5.11 Non-metric multidimensional scaling plot showing the effect of five disturbance regimes on the composition of regenerating nanophanerophytes for a) all sites, and b) all sites except U2, which had more species than other sites.

SIMPER analysis showed that of the 62 regenerating nanophanerophytic species, two species (*Phyllanthus calycinus* and *Opercularia echinocephela*) were regularly recorded in forest gap logged in 1990 and forest selectively logged prior to 1950 but were not recorded in forest that was burnt in 2000 ('burnt 2000' and 'gap logged/burnt 2000' treatments) (Table 5.11; Table 5.12). In contrast, one species (*Petrophile striata*) were more frequently recorded in forest burnt in

2000 than in forest gap logged in 1990 and forest selectively logged prior to 1950. Despite a lack of statistical difference ($p = 0.09$), small changes in the composition of regenerating nanophanerophytes were apparent between the two treatments representing forest burnt in 2000 (Table 5.10; Fig. 5.11). *Macrozamia reidlei* and *Leucopogon oxycedrus* were found in gap logged /burnt forest, but not in unlogged/burnt forest (Table 5.11; Table 5.12). Conversely, *Isopogon sphaerocephalus* and *Hakea ruscifolia* were more commonly recorded in unlogged/burnt forest than gap logged/burnt forest (Table 5.11; Table 5.12). Undisturbed forest did not have a statistically different composition to either recently burnt treatments because undisturbed forest contained the three species (*Petrophile striata, Isopogon spaerocephalus* and *Hakea ruscifolia*) that were more commonly found in recently burnt forest than in forest gap logged in 1990 or forest selectively logged prior to 1950 (Table 5.12).

Table 5.11 The regenerating nanophanerophytes that contributed to differences found between paired treatments that were identified as containing different species composition by ANOSIM analysis (Table 5.10a). Species are grouped within the 'treatment' they were more commonly encountered.

Treatments	**Species Contributing to Difference (% Contribution to Dissimilarity)**
Burnt 2000	*Hakea ruscifolia* (7.1)
	Isopogon sphaerocephalus (6.2)
	Petrophile striata (5.8)
Gap logged 1990	*Leucopogon oxycedrus* (5.7)
	Leucopogon propinquis (5.4)
	Phyllanthus calycinus (5.4)
	Macrozamia riedlei (5.1)
	Opercularia echinocephela (5.0)
Burnt 2000	*Hakea ruscifolia* (7.1)
	Isopogon sphaerocephalus (5.9)
	Petrophile striata (5.7)
Selectively Logged <1950	*Leucopogon oxycedrus* (7.1)
	Phyllanthus calycinus (7.1)
	Macrozamia riedlei (5.8)
	Opercularia echinocephela (5.8)
	Leucopogan verticillatus (5.8)
	Leucopogan pulchellus (5.2)
Gap Logged / Burnt 2000	*Petrophile striata* (7.4)
Gap logged 1990	*Phyllanthus calycinus* (7.4)
	Leucopogon propinquus (6.6)
	Opercularia echinocephela (5.5)
Gap logged / Burnt 2000	*Petrophile striata* (6.7)
Selectively Logged < 1950	*Phyllanthus calycinus* (6.7)
	Opercularia echinocephela (6.0)
	Leucopogon pulchellus (5.3)

Table 5.12 The obligate seeding nanophanerophytes recorded in five treatments representing different disturbance regimes ('Un' – Undisturbed, 'SL 50' – Selectively Logged < 1950, 'GL 90' – Gap Logged 1990, 'GLB' – Gap Logged/ Burnt 2000, 'B' – Burnt 2000). Numbers represent the number of sites that species were recorded.

FAMILY	SPECIES	Un	SL 50	Gl 90	GLB	B
APIACEAE	*Xanthosia atkinsoniana*	1	0	0	1	1
CASUARINACEAE	*Allocasuarina fraseriana*	3	3	3	2	3
DASYPOGONCEAE	*Kingia australis*	1	0	0	1	0
DILLENACEAE	*Hibbertia amplexicaulis*	3	2	3	4	2
	Hibbertia cuneiformis	1	0	0	0	0
	Hibbertia serrata	1	0	0	0	0
	Hibbertia commutata	4	4	4	4	3
	Hibbertia hypericoides	4	4	3	4	4
	Hibbertia pachyrrhiza	3	3	4	4	4
EPACRIDACEAE	*Astroloma ciliatum*	2	2	2	3	1
	Astroloma glaucescens	1	0	1	1	0
	Astroloma pallidum	4	3	3	4	4
	Leucopogon oxycedrus	1	4	4	2	0
	Leucopogon propinquus	2	1	4	1	0
	Leucopogon pulchellus	3	3	2	1	1
	Leucopogon verticillatus	2	3	0	2	0
	Leucopogon sp.1	1	0	0	0	0
EUPHORBIACEAE	*Phyllanthus calycinus*	2	4	4	0	0
FABACEAE	*Labichea punctata*	1	2	1	2	0
	Daviesia cordata	0	0	0	1	1
	Daviesia decurrens	3	2	2	3	2
	Daviesia polyphylla	0	0	0	1	1
	Daviesia preissii	0	1	1	1	2
	Daviesia flexuosa	0	0	0	0	1
	Hovea elliptica	2	0	0	0	0
	Bossiaea ornata	4	4	4	4	4
	Hovea chorizemifolia	4	4	3	4	4
GOODENIACEAE	*Lechenaultia biloba*	3	4	4	4	4
LAMIACEAE	*Hemiandra pungens*	1	0	0	0	0
MYRTACEAE	*Agonis linearifolia*	1	0	0	0	0
	Astartea fascicularis	1	0	0	0	0
	Baeckia camphorosmae	1	0	0	0	2
	Calothamnus quadrifidus	0	0	0	0	1
	Corymbia calophylla	4	4	4	4	3
	Eucalyptus marginata	4	4	4	4	4
	Hypocalymma angustifolium	1	0	0	2	1
	Melaleuca incana	1	0	0	0	0
PROTEACEAE	*Adenanthos barbigerus*	3	4	4	4	4
	Banksia grandis	4	3	2	4	3
	Banksia littoralis	1	0	0	0	0
	Grevillea pilulifera	2	1	0	0	1
	Grevillea synapheae	3	4	4	4	4
	Grevillea wilsonii	0	1	0	0	1
	Hakea amplexicaulis	3	2	3	4	4
	Hakea cyclocarpa	1	1	1	0	2
	Hakea lissocarpha	2	2	3	2	1
	Hakea ruscifolia	1	0	1	1	4
	Isopogon sphaerocephalus	2	0	0	1	3
	Isopogon teretifolius	0	1	0	0	2
	Persoonia elliptica	0	0	0	0	1
	Persoonia longifolia	3	4	3	1	2
	Petrophile seminuda	0	1	0	0	0
	Petrophile striata	1	0	0	4	3
	Petrophile sp.1	0	0	0	0	1
	Synaphea sp. 1	0	0	0	0	1
	Synaphea spinulosa	1	1	0	0	1
	Synaphea petiolaris	0	2	1	0	0
RUBIACEAE	*Opercularia echinocephala*	2	3	3	0	0
RUTACEAE	*Eriostemon spicatus*	1	0	0	0	2
XANTHORRHOEACEAE	*Xanthorrhoea gracilis*	4	4	4	4	4
	Xanthorrhoea preissii	4	3	4	4	4
ZAMIACEAE	*Macrozamia riedlei*	3	3	3	3	0
Total Number of Species / Treatment		50	37	33	37	42

The vast majority of regenerating nanophanerophytes were patchily distributed among sites. There were 26 species (42 % of the regenerating nanophanerophytic community) that were recorded in less than four sites (Table 5.12). Of these 26 species, 15 were found in undisturbed forest, and 15 in unlogged forest that had been burnt in 2000. This is more than the number of these 'uncommonly' recorded regenerating nanophanerophytes recorded in forest gap logged/ burnt in 2000 (n = 6 spp.), gap logged in 1990 (n = 2 spp.) and selectively logged prior to 1950 (n = 5 spp.).

5.3.5 Regenerating Hemicryptophytes

Species Richness

Between three and nine species of regenerating hemicryptophyte were recorded across sites ($\bar{x}$ = 5.9 SE ± 0.4 species per site). There were a similar number of species among all treatments (Fig. 5. 12). Undisturbed forest had smaller variation than found in other treatments (Fig. 5.12). No statistical difference in the number of regenerating hemicryptophytic species were found between all treatments (Table 5.13). When the was low statistical power in the ANOVA model (power = 0.15) was taken into account, there were also no differences among treatments ($p > 0.1$) (Table 5.13). *Post hoc* analysis revealed that 32 sites per treatment would have provided the power to detect any possible impact (power > 0.8).

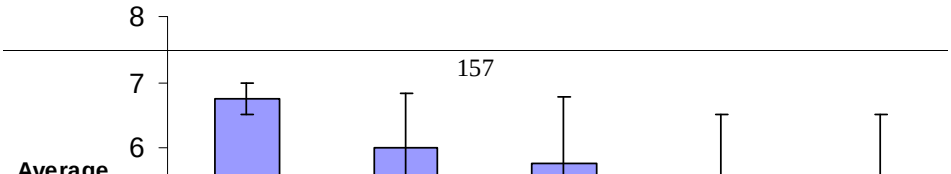


Fig. 5.12 The average (± SE) species richness of regenerating hemicryptophytes per site for five treatments representing different disturbance regimes. There were no significant differences between treatments ($p > 0.1$).

Table 5.13 One-way ANOVA and power analysis testing whether there are differences in the species richness of regenerating hemicrytophytes between treatments representing different disturbance regimes.

	Sum of Squares	Degrees of Freedom	Mean Square	F-ratio	P Value	POWER
Between Groups	0.22	4	0.05	0.56	0.70	0.15
Within Groups	1.45	15	0.10			
Total	1.66	19				

Species Composition

Variations in the composition of regenerating hemicryptophytes between sites were not attributable to past management practices (Global $R = 0.119$, $p = 0.11$). There were no discernible differences in composition between sites on the ordination (Fig. 5.13). This may be because five species (*Pentapeltis peltigera, Drosera erythrorhiza, Drosera pallida, Conostylis caricina* and *Patersonia occidentalis*) comprise almost 70% of the regenerating hemicryptophytes records (Table 5.14). Eight other species were recorded in less than 4 sites. These species were distributed relatively evenly throughout the five treatments (undisturbed = 2 spp., selectively logged < 1950 = 2 spp. , gap logged 1990 = 1 spp., gap logged/burnt 2000 = 2 spp., burnt 2000 = 4 spp.)

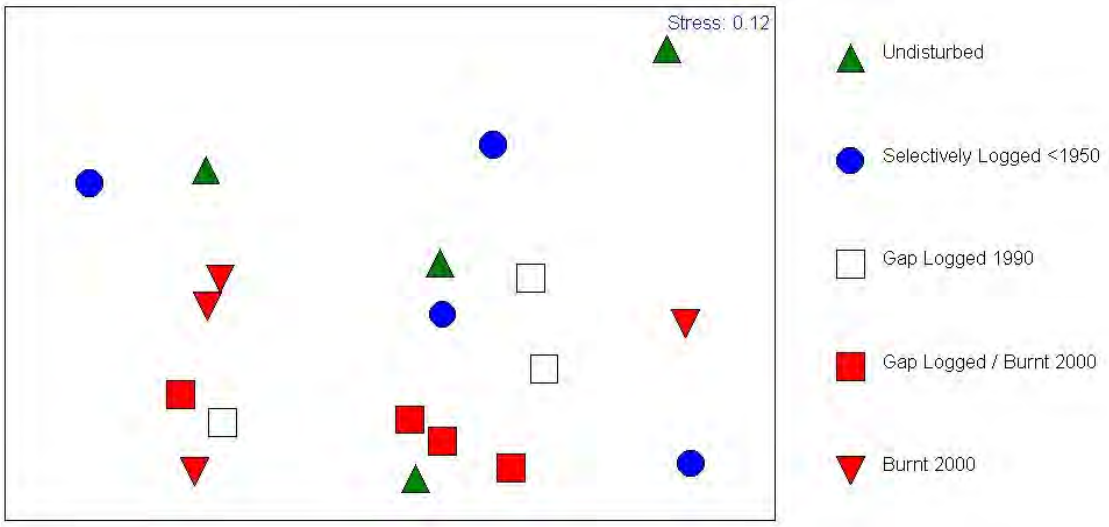


Fig. 5.13 Non-metric multidimensional scaling plot showing the effect of five disturbance regimes on the composition of regenerating hemicryptophytes.

Table 5.14 The regenerating hemicryptophytes recorded in five treatments representing different disturbance regimes ('Un' – Undisturbed, 'SL 50' – Selectively Logged < 1950, 'GL 90' – Gap Logged 1990, 'GLB' – Gap Logged/ Burnt 2000, 'B' – Burnt 2000). Numbers represent the number of sites that species were recorded.

FAMILY	SPECIES	Un	SL 50	Gl 90	GLB	B
APIACEAE	*Pentapeltis peltigera*	3	4	4	4	4
DROSERACEAE	*Drosera erythrorhiza*	3	3	3	3	1
	Drosera gigantea	0	0	0	0	1
	Drosera menziesii	0	1	0	0	0
	Drosera pallida	3	2	3	3	1
	Drosera stolonifera	1	2	3	0	1
	Drosera sp. 1	0	0	0	1	0
	Drosera sp. 2	0	0	0	0	1
FABACEAE	*Sphaerolobium medium*	2	0	0	0	1
HAEMODORACEAE	*Anigozanthos manglesii*	0	0	1	1	0
	Conostylis caricina	4	3	4	4	4
	Conostylis serrulata	3	1	0	0	1
	Conostylis sp.1	0	1	0	0	1
IRIDACEAE	*Orthrosanthos laxus*	1	0	0	0	0
	Patersonia pygmaea	1	1	0	3	0
	Patersonia occidentalis	3	3	4	4	4
PHORMIACEAE	*Dianella revoluta*	3	3	1	0	1
Total Number of Species / Treatment		11	11	8	8	12

Despite there being no statistical difference between treatments for the entire regenerating hemicryptophytic community, some species appeared to be affected by different management practices. *Dianella revoluta* var. *divaricata* was regularly found in undisturbed forest and forest selectively logged prior to 1950, but not in any other treatment (Table 5.14). *Conostylis serrulata* was found regularly in undisturbed forest, but not in forest gap logged in 1990 or 2000

(Table 5.14). *Patersonia pygmaea* was more commonly recorded after gap logging and fire than in unlogged, burnt forest (Table 5.14).

4.3.6 Regenerators - Cryptophytes

Species Richness

Although 58 regenerating cryptophyte species were recorded at Mt Dale, the average species richness of this group was low for all sites ($\bar{x}$ = 8.4 SE ± 0.5, species per site, range = 3-13 species per site). There were no statistical differences in the number of regenerating cryptophyte species found between treatments (Table 5.15; Fig. 5.14). When low statistical power in the ANOVA model (power = 0.39) was taken into account, there was also no differences among treatments ($p > 0.1$) (Table 5.15). *Post hoc* power analysis revealed that twice the number of sites (i.e., 8 sites/ treatment) would be required to detect differences using standard levels of significance (power > 0.8).

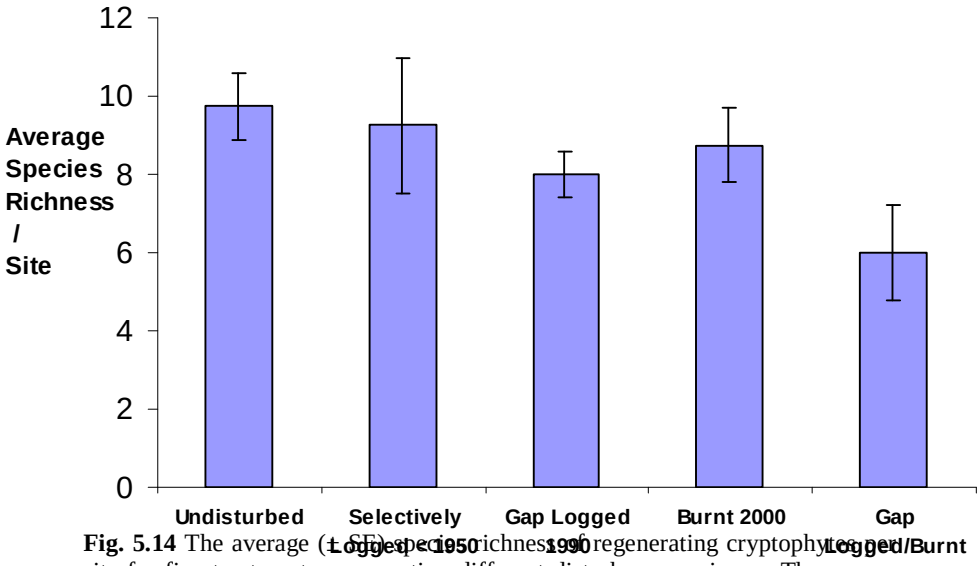


Fig. 5.14 The average (± SE) species richness of regenerating cryptophytes per site for five treatments representing different disturbance regimes. There were no significant differences between treatments.

Table 5.15 One-way ANOVA and power analysis testing whether there are differences in the species richness of regenerating crytophytes between treatments representing different disturbance regimes.

	Sum of Squares	Degrees of Freedom	Mean Square	F-ratio	P value	POWER
Between Groups	0.71	4	0.18	1.76	0.19	0.39
Within Groups	1.51	15	0.10			
Total	2.22	19				

Species Composition

No significant differences in the composition of regenerating cryptophytes could be attributed to past or present management practices (Global R = 0.072, p = 0.11). This lack of statistical difference was in part due to communities of regenerating cryptophytes being dominated by patchily distributed species which created considerable variation between sites (Fig. 5.15). Of the 58 species identified, only five (9 %) species were recorded in all treatments (*Lepidosperma pubisquameuem, Lepidosperma tenue, Lomandra micranta, Lomandra* sp. 1 and *Dryandra lindleyana*) (Table 5.16). Forty seven (81 %) species were recorded in less than 4 sites (Table 5.16). Of the 47 species, undisturbed forest and selectively logged forest (n = 16 spp. / treatment) had more species than gap logged forest in 1990 (n = 6 spp.) (Table 5.16). Unlogged forest that was burnt

in 2000 had more of these uncommon species (n= 13 spp.) that forest that gap logged before being burnt in 2000 (n = 9 spp.) (Table 5.16).

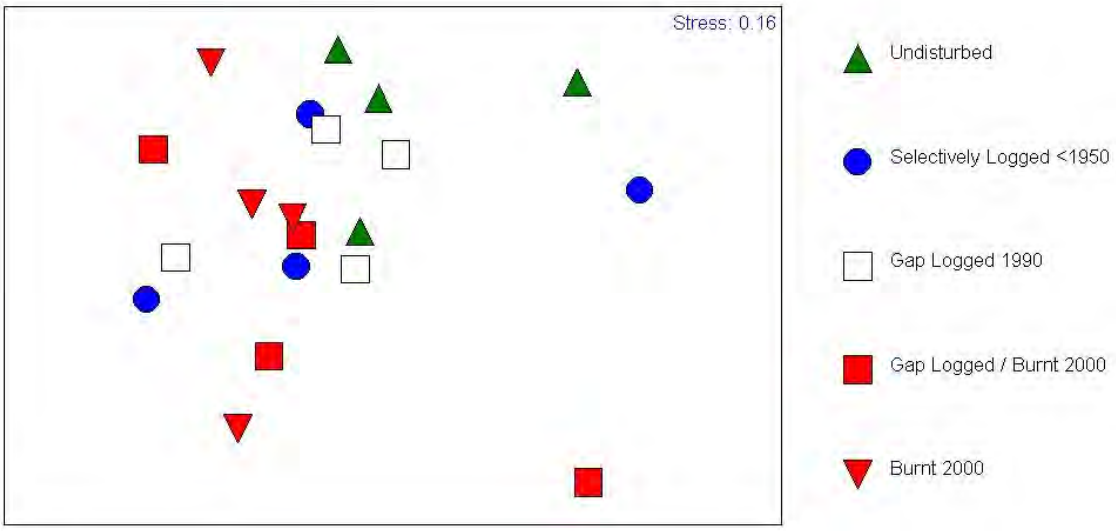


Fig. 5.15 Non- metric multidimensional scaling plot showing the effect of five disturbance regimes on the composition of regenerating cryptophytes.

Table 5.16 The regenerating cryptophytes recorded in five treatments representing different disturbance regimes ('Un' – Undisturbed, 'SL 50' – Selectively Logged < 1950, 'GL 90' – Gap Logged 1990, 'GLB' – Gap Logged/ Burnt 2000, 'B' – Burnt 2000). Numbers represent the number of sites that species were recorded.

FAMILY	SPECIES	Un	SL 50	Gl 90	GLB	B
ANTHERICACEAE	*Chamaescilla corymbosa*	0	0	0	0	1
	Thysanotus patersonii	1	0	0	0	0
	Thysanotus sparteus	1	0	0	0	0
	Thysanotus sp. 1	0	1	0	0	0
	Thysanotus sp. 2	0	0	0	0	1
	Thysanotus sp. 3.	0	0	0	0	1
COLCHICACEAE	*Burchardia multiflora*	0	0	0	0	2
	Burchardia umbellata	2	1	2	1	0
CYPERACEAE	*Lepidosperma pubisquameum*	2	1	2	1	3
	Lepidosperma tenue	3	3	4	3	4
	Lepidosperma tetraquetrum	1	0	0	0	0
	Lepidosperma sp. 1	0	1	0	0	0
	Lepidosperma sp. 2	1	0	0	0	0
	Lepidosperma sp. 3	0	1	0	0	0
	Lepidosperma. sp. 4	0	0	1	1	0
	Lepidosperma sp. 5	1	0	0	0	0
	Mesomelaena tetragona	1	0	0	0	1
	Mesomelaena sp. 1	0	0	0	2	0
	Mesomelaena sp. 2	0	0	0	0	1
DASYPOGONCEAE	*Lomandra integra*	0	0	0	1	0
	Lomandra micrantha	4	3	3	1	3
	Lomandra preissii	2	0	0	0	0
	Lomandra sp.1	1	2	2	2	3
HAEMODORACEAE	*Haemodorum laxum*	0	0	0	1	0
	Haemodorum simulans	1	0	0	0	0
	Haemodorum sp. 1	0	0	0	1	2
	Haemodorum sp. 2	0	1	0	1	1
	Haemodorum sp. 3	1	0	1	1	1
	Haemodorum sp. 4	0	0	0	0	1
	Haemodorum sp. 5	0	0	0	0	1
ORCHIDACEAE	*Burndettia nigricans*	3	2	3	0	0
	Caladenia flava	0	2	3	0	0
ORCHIDACEAE	*Caladenia deformis*	0	0	0	1	1
	Caladenia longicauda	0	1	0	0	0
	Elythranthera brunonis	0	0	1	0	1
	Pterostylis aspera	0	1	0	0	0
	Pterostylis barbata	0	1	0	0	0
	Pterostylis nana	0	1	0	0	0
	Pterostylis recurva	2	1	1	0	0
	Pterostylis sp. 1	1	1	0	0	0
	Thelymitra crinita	0	1	0	0	0
PITTOSPORACEAE	*Pronaya fraseri*	1	1	0	0	0
	Sollya heterophylla	0	0	1	1	0
POLYGALACEAE	*Comesperma virgatum*	0	0	0	0	1
PROTEACEAE	*Dryandra bipinnatifida*	0	1	0	0	0
	Dryandra lindleyana	4	4	4	4	4
RANUNCULACEAE	*Clematis pubescens*	0	1	0	0	0
RESTIONACEAE	*Desmocladus flexuosus*	0	2	1	0	1
	Desmocladus sp. 1	2	0	1	0	0
	Desmocladus sp. 2	0	1	0	0	0
	Desmoscladus sp. 3	0	0	1	0	1
	Lepidobolus chaetocephalus	1	0	0	0	0
	Meeboldina coangustata	1	0	0	0	0
	Restionaceae sp. 1	0	0	1	0	0
	Restionaceae sp. 2	1	1	0	0	0
	Restionaceae sp. 3	0	0	0	1	0
	Restionaceae sp.4	1	1	0	0	0
	Restionaceae sp. 5	0	0	0	1	0
Total Number of Species / Treatment		**24**	**26**	**17**	**17**	**21**

5.4 Discussion

The composition of understorey plants changed dramatically among sites at Mt Dale. Over half of the 182 species were recorded at less than four sites, where as only five were recorded at all sites. Much of this 'natural' variation is caused by small changes in topography, available moisture and edaphic properties (Havel 1975b; McCutcheon 1980; Strelein 1988; Hopper 1992; Hopper *et al.* 1996; Goia & Pigott 2000). However, small changes in the diversity (species richness and composition) of understorey communities at Mt Dale could be attributed to fire and logging disturbance. With respect to fire, less species were recorded in the two treatments disturbed by fire in 2000 ('gap/logged burnt 2000' and 'burnt 2000') than in undisturbed forest. Studies in other low rainfall zones in the jarrah forest show that species richness will recover seven years after fire (Burrows & Wardell-Johnson 2003). With respect to logging, less species were recorded in forest gap logged in 1990 than in undisturbed forest. These changes show that gap logging disturbs plant diversity for a longer period of time than fire disturbs plant diversity. The diversity of understorey plants was similar in forest selectively logged prior to 1950 and undisturbed forest, which implies that plant communities have recovered from previous selective logging disturbance.

5.4.1 The Impact of Fire in Unlogged and Logged Forest

The immediate impact of a moderate intensity fire is the destruction of almost all understorey vegetation. Four months after the '2000' fire at Mt Dale, less species were recorded in the two treatments that were disturbed by this fire ('gap logged/burnt 2000' and 'burnt 2000') than the three treatments that had not been burnt since prior to 1995. There were predictable differences in the response between those species that regenerate from buds (regenerators) and those that

germinate from seed after fire (obligate seeders). The species richness and composition of renegenerating nanophanerophytes, hemicryptophytes and cryptophytes were similar between unlogged forest burnt in 2000 and unlogged forest burnt prior to 1995, indicating the diversity of these species had regenerated from perenniating tissue protected from fire. In contrast, fewer species of obligate seeding nanophanerophytes and hemicryptophytes were recorded in the unlogged forest that was burnt in 2000 ('undisturbed') than in unlogged forest that was burnt prior to 1995 ('burnt 2000'). The recovery of obligate seeding species after fire is dependent on seed germination, which in jarrah forest has been shown to be dependent on many factors including available soil moisture and the season of the last fire (Burrows & Wardell-Johnson 2003; Grant 2003). Interestingly, more obligate seeding species germinate after autumn fires (such as the 2000 fire at Mt Dale) than after spring fires (Grant & Loneragan 2001). This is thought to occur because: i) most species germinate in autumn possibly due to higher soil moisture following winter rains (McCaw 1988); ii) autumn fires tend to have a higher intensity than spring fires, which leads to greater consumption of potentially inhibiting litter layer and higher establishment rates (Bond & van Wilgen 1996); and iii) competition from regenerating species after spring fires is greater than after autumn fires due to the increased time available for reprouters to grow before germination of seeding species commences (Hobbs & Atkins 1990).

There were small differences in the number and composition of species between the two treatments that were burnt in 2000. The species richness of obligate seeding nanophanerophytes and hemicryptophytes was lower in unlogged forest that had been burnt in 2000 than in forest that had been gap logged and burnt in

2000. There were also small differences in the composition, but not species richness, of regenerating plants between these two treatments. Species including *Macrozamia reidleii* (regenerating nanophanerophyte) and *Leucopogon oxycedrus* (regenerating nanophanerophyte) were recorded more frequently in forest that was gap logged and burnt in 2000 than unlogged forest that burnt in 2000. In contrast, species including *Isopogon sphaerocephalus* (regenerating nanophanerophyte), *Hakea ruscifolia* (regenerating nanophanerophyte) and *Conostylis serrulata* (regenerating hemicryptophyte) were more commonly recorded in 'burnt 2000' sites than 'gap logged/ burnt 2000' sites. In addition, unlogged forest burnt in 2000 had more of the species that were considered 'uncommon' (i.e., those that were recorded at less than four sites) (34%) than forest that was gap logged and burnt in 2000 (22%). These results show that a small proportion of the species in the understorey plant community respond differently to fire if this disturbance is preceded by logging.

The differences in species assemblages between the burnt treatments could occur for several reasons. The first reason is that gap logging affected plant communities prior to fire disturbance. Heavy machinery used in logging operations can disturb near surface soil profiles and destroy regenerative parts of plants (Burrows *et al.* 2002b). After gap logging, over 60% of the topsoil in the patch can be disturbed by heavy machinery (Burrows *et al.* 2002b). Similar losses in regenerating species after mining operations in jarrah forest have been attributed to the low seed production and low seed viability of regenerating species (Grant & Loneragan 1999a,b). Furthermore, logging disturbance may cause increased herbivory which can reduce the number of viable seeds and the survival success of germinating plants (Burrows *et al.* 2002b; Koch *et al.* 2004).

Moreover, the loss of canopy cover (Chapter 4) is likely to result in reductions in soil moisture because of the increased incidence of solar radiation on the soil surface. In other forests, loss of soil moisture negatively affects the establishment and growth of seedlings (Beatty 1984; Gray & Spies 1997). Any of these changes could have resulted in a different assemblage of resprouting and seeding plants after gap logging, and caused the differences found between unlogged/burnt forest and gap logged/burnt forest.

The second possibility is that fire properties were different between the two treatments. The increased volume of woody debris that exists after logging operations presents a vastly different fuel complex than that which accumulates naturally in an unlogged forest (Burrows *et al.* 2001; Chapter 4). This altered fuel complex is likely to have increased the intensity of fire (Hobbs 2003). Changes in fire intensity has been shown to influence the germination and regeneration of many understory species (e.g., Beadle 1940; Peet 1971; Shea *et al.* 1979; Burrows 1999; Burrows & Wardell-Johnson 2003). The composition of the life forms that germinate from a seed bank (Smith *et al.* 2000) as well the composition of regenerating species (Peet & McCormick 1971) can depend on the intensity of a particular fire. For example, thickets of leguminous vegetation have often been observed following high intensity fires but are rarely associated with low or moderate intensity fires in jarrah forest (McCaw 1988). Changes in fire intensity not only change the composition of species that immediately germinate or resprout after a given fire, but also can affect the density of arbuscular mycorrhizal fungi at the soil surface, which are hypothesised as important for seedling establishment (Pattinson *et al.* 1999). Therefore, gap logging may influence the composition of understorey species by increasing the

intensity of fires in jarrah forest. It is likely that a combination of these factors has caused different plant communities to occur in unlogged/burnt forest compared with that in gap logged/burnt forest.

5.4.2 Plant Diversity 10 years after Gap Logging

There were less understorey plant species in forest disturbed by gap logging in 1990 than undisturbed forest. This could not be associated to a low representation or absence of any particular guild. However, trends suggest that regenerating species appear more susceptible to gap logging than obligate seeders. Of the uncommonly recorded species (i.e., those species recorded at less than four sites), more regenerating nanophanerophytes (n = 12 species) and cryptophytes (n = 10 species) were recorded across sites in undisturbed forest than forest gap logged in 1990. These trends are weak, representing less than 20 % of all species of regenerating nanophanerophyte (19 %) and regenerating cryptophytes (17 %). However, these reductions should be noted because similar reductions in regenerating species have been observed following logging disturbance in other ecosystems (Haeussler *et al.* 2002) and mining disturbance in the jarrah forest (Grant & Loneragan 1999a,b).

Contemporary logging may cause ongoing adverse impacts on regenerating plants. This is because following disturbance, the jarrah forest understorey communities tend to follow the 'initial floristic composition' model of successional development (Grant & Koch 2003). This model predicts that the composition of the flora immediately following disturbances determines future shifts in dominance, with various species successively becoming more dominant as their life-history characteristics and associated life-forms are exhibited with

the passage of time (Egler 1954). If a large proportion of regenerating plants on a site is destroyed by heavy machinery, then populations of resprouting species are expected to recover slowly because they produce lower quantities of viable seeds than obligate seeding species (Bell *et al.* 1987). Burrows *et al.* (2002b) suggested that recolonisation, and therefore recovery, of some eliminated species is likely to be very slow after logging in jarrah forest. Studies that have investigated similar intensive logging regimes in other temperate forests have shown that some regenerating plant communities can be affected for at least 30 years (Halpern *et al.* 1992; Hickey 1994; Packham *et al.* 2002).

5.4.3 Plant Diversity 50 years after Selective Logging

There was no difference in diversity (species richness and composition) of all guilds between sites in forest selectively logged prior to 1950 and undisturbed forest. This implies that forest selectively logged prior to 1950 has recovered from early logging disturbance. The similarity in plant communities between unlogged forest and forest selectively logged prior to 1950 has important ramifications for management. The entire production forest estate in Western Australia now comprises forest that has previously been logged. Much of the jarrah forest in this estate was selectively logged prior to 1950. By showing that forest selectively logged prior to 1950 in the production forest estate has the same diversity level of biodiversity as unlogged forest in reserves, managers can more confidently maintain diversity of understorey plants in production forests. A landscape mosaic can be designed so that forest selectively logged prior to 1950 and undisturbed can be managed to buffer against any deleterious outcomes of contemporary logging.

5.4.4 Future Research

This research has raised several important points that should be recognised in future studies. The first is that all ANOVAs in this research that assessed species richness contained low statistical power. Low statistical power may arise either because there is no difference between the treatments or because the experiment lacked the sensitivity to detect an impact that actually occurred (Peterman 1990). For obligate seeding nanophanerophytes and obligate seeding hemicryptophytes, it was suspected that the experiment lacked the sensitivity to detect an impact that actually occurred. *Post hoc* power analysis showed that at least eight sites per treatment were required to adequately assess whether the null hypothesis could be rejected by the ANOVA. Future research should therefore attempt to have at least eight sites for each treatment if using the same methodology as was used in this research.

The second important outcome of this research was that the two measures of species diversity (α and β diversity) were not equivalent. For example, the species richness of regenerating phanerophytes was same between burnt and unburnt treatments (Fig 5.10), but there were large differences in the composition of this guild between these treatments (Fig. 5.11). This result has enormous ramifications for managers because it shows that measuring the recovery of species richness after a disturbance does not accurately determine whether the pre-existing community has recovered from the disturbance (as also shown by Summerville & Crist 2002). Future research that gauges how disturbance affects plant diversity in jarrah forest must assess whether the composition of species has been affected before it can be confidently concluded that the disturbance has had no impact on plant diversity.

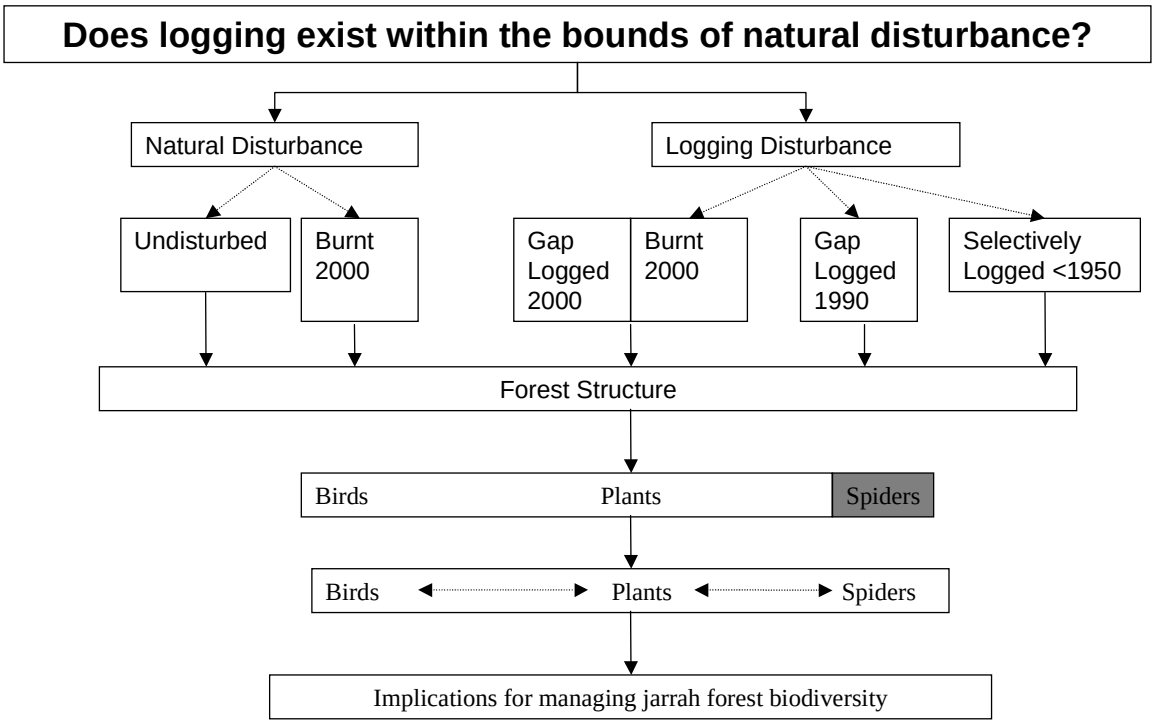
The third significant finding is the recognition that particular plant guilds appear more susceptible to logging disturbance than others. At Mt Dale, regenerating species appear to be the more negatively affected than obligate seeding species following gap logging. In a follow-up study to this research, Slee's (2003) honours research supported this finding by revealing that there were more species of regenerating cryptophytes and hemicryptophytes in unlogged forest than in forest that had been selectively logged 19-34 years before. Given that these guilds appear to be negatively affected by logging disturbance, future research could focus on assessing how regenerating plants re-establish in sites that have been previously affected by logging.

5.4.5 Conclusion

The understorey plant community in the jarrah forest at Mt Dale was extremely heterogeneous over small spatial scales. Small trends in species richness and composition could be attributed to logging and fire disturbance. There were less obligate seeding species in forest burnt in 2000 than forest burnt prior to 1995. Furthermore, the richness (obligate seeders) and composition (regenerators) of species in unlogged forest that had been burnt in 2000 was different to forest that had been gap logged and burnt in 2000. With respect to the treatments that had not been disturbed by fire in 2000, less species were recorded in forest gap logged in 1990 than in undisturbed forest. As plant diversity recovers within seven years of fire, this result shows that gap logging affects understorey plant communities for a longer period of time than fire. The similarity in the diversity of plants in unlogged forest and forest selectively logged prior to 1950 shows that the plant community has recovered from selective logging disturbance.

CHAPTER 6

Ground Spiders



Chapter 6

6.1 Introduction

Invertebrates comprise most of the biodiversity within eucalypt forests (Recher *et al.* 1996). They are the principal consumers of forest primary production and are central to ecosystem functioning in jarrah forests (Majer & Abbott 1989; Friend 1995; van Huerck *et al.* 1999). Studying forest invertebrates is therefore likely to provide a basis on which to assess the long-term sustainability of forests, as producers of wood and fibre, and as repositories of biodiversity. Among invertebrates, the third most biodiverse and abundant taxon is Araneae (spiders) (Brunet 1994). Spiders are at the apex of the invertebrate food pyramid and forage primarily on other invertebrates (Enders 1975; Uetz 1976; Hurd & Fagan 1992; Jennings *et al.* 1988). As such, spiders are considered useful biodiversity indicators because they are capable of reflecting ecological impacts at lower levels in the food chain (Brennan 2002).

In jarrah forest, ground-dwelling spiders have been used to evaluate mine site rehabilitation (e.g., Mawson 1986; Simmonds *et al.* 1994; Brennan 2002), prescribed burning (e.g., van Huerck *et al.* 1998) and logging (e.g., Abbott *et al.* 2002). These studies have revealed that this fauna is extremely diverse, relatively easy to sample, and responsive to many types of disturbance in jarrah forest (Main 2001; Brennan *et al.* 2004). In other temperate forests, investigations that have compared the impact of fire with logging have successfully used spiders as indicator taxa (Huhta 1971; Buddle *et al.* 2000). Consequently, ground dwelling spiders were selected to assess whether there are differences between fire and logging disturbance in jarrah forest.

There have been several studies that have assessed the impact of fire on ground dwelling spiders in jarrah forest. Those studies that sorted invertebrates to the ordinal rank show that spider density can be affected for at least 2 years following fire (Majer 1984; Abbott 1984a; Abbott *et al.* 1984). Three other studies identified ground spiders to morpho-species and therefore were able to assess whether fire affected spider species richness. Springett (1979) looked at the meso- and microarthropod faunas between unburnt and burnt (moderate intensity fire in March 1975) forest, and showed burning reduced spider species richness for at least four months after fire. Van Huerck *et al.* (1998) looked at pyric succession following spring and autumn prescribed burnt in the central jarrah forest. Their study showed that spider species richness declined significantly for two years after fire irrespective of the season in which the fire occurred. Changes in the composition of spider communities were not examined.

The third study that identified spiders to morpho-species is the most comprehensive assessment of the response of spider communities to a single fire event. Brennan (2002) described spider succession following spring fires in the jarrah forest around Jarrahdale, a town located in the jarrah northwest ecosystem. Spiders were sampled by pitfall trapping and vacuum sampling from understorey vegetation at sites representing a chronosequence of increasing age classes since burning (zero, three, six, and nine years). Immediately following fire, the mean species richness reduced significantly but returned after three years. However, changes in taxonomic composition (both at familial and specific ranks) were still apparent three years after fire. It is presumed they return to pre-fire levels after 6

years because taxonomic composition was equivalent in the two older treatments (6 years post fire and 9 years post fire).

In contrast to fire, very little is known of the effect of logging on spider diversity in jarrah forest. Only two studies have assessed how spiders are affected by this disturbance, and both were part of the Kingston study. The first study placed spiders, with other ground arthropods, into broad taxonomic groups (ordinal level) (Strehlow *et al.* 2002). This study assessed the short term impact of three treatments of logging on ground-dwelling invertebrates: gap logging with no habitat trees (clearfelling); gap logging with habitat trees; and shelterwood logging. The abundance of spiders was not affected by logging. However, this research had low statistical power and the acceptance of the null hypothesis should be treated cautiously (power = 0.33, Table 1 in Strehlow *et al.* 2002). It is also important to note that their study did not assess the entire disturbance caused by contemporary logging operations within jarrah forest, as no post-logging burn was conducted.

The second study that assessed the impact of logging in the Kingston forest block identified ground dwelling spiders to morpho-species (Abbott *et al.* 2002). Their research compared spider diversity (as well as that of cockroaches, crickets and grasshoppers) between Gap release (logged 1995/96, burnt 1996), Coupe Buffer (sites located in a buffer between gap logging sites and burnt in 1996) and external controls (unlogged for at least 50 years and unburnt for at least 10 years). This study showed that immediate decreases in spider species richness were rapidly reversed (*c.* 18 months). Ordinations (nMDS) revealed that the community structure in both 'treatment' and control sites at the end of the study

was different from the community structure at the beginning of the study, indicating the overriding importance of climatic variation. However, no statistical assessment of possible compositional changes caused by logging was undertaken.

Measuring changes in composition is essential when assessing how spider diversity is affected by logging or fire disturbance. In other forests, compositional changes in spider communities remain for many years following logging or burning; and this is after species richness returned to pre-disturbance levels (Huhta 1971; Coyle 1981; McIver *et al.* 1992; Buddle *et al.* 2000; Willett 2001). These compositional changes are usually detected by placing spiders within guilds based on how they capture prey (Uetz *et al.* 1999). In Europe, where the foraging behaviour of spiders is well known, logging differentially affects species with different prey-capture behaviours (e.g., Coyle 1981; McIver *et al.* 1990; Willet 2001). This approach was applied to the spider community at Mt Dale, although it should be treated cautiously because the behaviour of many Australian species is unknown. Therefore, this research aims to add to the findings of the Kingston study by placing ground spiders within guilds and addressing the following three questions:

1) Are there differences in the richness or assemblage of spider species within different guilds in forest recently disturbed by fire compared with forest recently disturbed by gap logging and fire?

2) Does the richness or assemblage of spider species within different guilds recover 10 years after gap logging?

3) Does the richness or assemblage of spider species within different guilds recover 50 years after selective logging?

6.2 Methods and Materials

6.2.1 Selecting Guilds

Spiders were differentiated into two foraging guilds based on the knowledge of family behaviour elsewhere in the world (as of Uetz 1977; Buddle *et al.* 2000):

1. '**Hunting**', or 'wandering', spiders are those that do not rely on silk for capturing prey. These spiders usually have good eyesight, and hunt by leaping, springing, running, or ambushing prey that comes within their range (Clyne 1969; Uetz 1976; Pajunen *et al.* 1995). After logging and fire, hunting spiders often quickly invade (e.g., Jennings *et al.* 1988; McIver *et al.* 1992; Buddle *et al.* 2000; Pajunen *et al.* 1995; Willet 2001). The ability of these spiders to successfully contend with, and even take advantage of, these disturbances is thought to occur for several reasons: i) hunters move readily between patches that contain favourable resources, ii) hunters use woody debris resulting from logging to shelter, and, iii) hunters predate upon the herbivores that increase as a result of regenerating vegetation following fire, and the decomposers that increase as a result of the accumulation of woody debris (Coyle 1981; McIver *et al.* 1992; Wise 1993).

2. '**Web building',** or 'web weaving'**,** spiders rely on webs to capture prey (Brunet 1994). Specific environmental attributes are required for the construction and maintenance of webs, and conduction of vibratory signals that alert the spider to captured prey (Wise 1993). The diversity of web building spiders has been related to forest structural attributes including leaf litter structure (Huhta 1971; Uetz 1979; Bultman & Uetz 1982; McIver *et al.* 1992), vegetation structure (Greenstone 1984; McIver *et al.* 1990), canopy cover (Coyle 1981; McIver *et al.* 1992; Pajunen *et al.* 1995), distribution of

overstorey trees (Willet 2001) and herb, lichen and moss cover (Pajunen *et al.* 1995; Buddle *et al.* 2000; Willet 2001). As logging can influence many of these attributes, reductions in the diversity of web-building spiders have often been associated with this disturbance (e.g., Koponen 1995; Pajunen *et al.* 1995; Buddle *et al.* 2000; Willett 2001).

Species that use other hunting techniques were not considered in this study.

6.2.2 Sampling Spiders

At Mt Dale, ground spiders were sampled using pitfall traps; a technique following the methods of Abbott *et al.* (2002). They successfully used pitfall traps to assess the short-term impact of logging on spider diversity in jarrah forest. This technique allows for simultaneous sampling across a range of sites, thereby reducing variation caused through sampling at different times and in different weather conditions (Majer 1997; Brennan *et al.* 1999). Although it is expected that pitfall traps will capture more mobile species (e.g., hunting spiders), this technique has also been successfully used in many forests worldwide to sample web building spiders (e.g., Jennings *et al.* 1988; Koponen 1995; McIver *et al.* 1992; Niemela *et al.* 1993; Pajunen *et al.* 1995; Willet 2001).

At each of the 20 sites, ten pitfall traps were placed along one 100m transect (Fig. 6.1). As pitfall trap location is usually best determined by a random stratified design (Uetz 1976), each pitfall trap was placed at a random location along a 10 m sub-section of transect (Fig. 6.1). This ensured that the entire length of a site was assessed. The transect had a randomly selected orientation (Fig. 6.1).

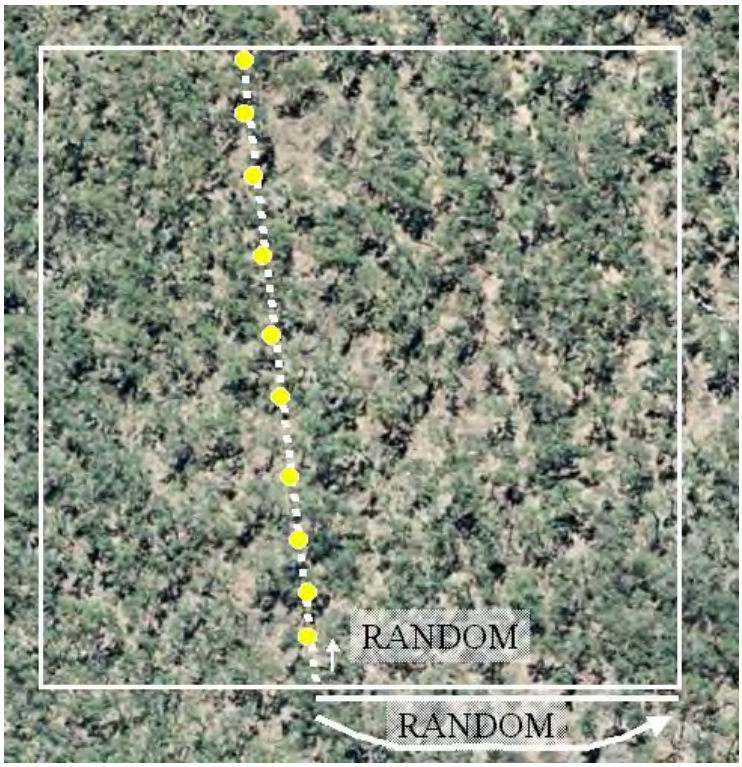


Fig. 6.1 An example of the layout of pitfall traps (yellow dots) for one site (as indicated by the white square). From a randomly selected corner, a random distance was measured to the start of a 100m transect line. One pitfall trap was placed at a random point within each 10 m subsection on the transect line.

Pitfall trap diameter is an important factor influencing the capture rate of arthropods (Abensperg-Traun & Stevens 1995). Following Brennan *et al.* (1999), who showed that traps less than 7cm in diameter may decrease the potential for capture, each pitfall trap consisted of a 20cm length of PVC pipe, 9cm in diameter, placed vertically into the ground. The sleeves of pitfall traps were buried with their top at ground level. To avoid 'digging in' effects (Greenslade 1973), each trap was installed at least a week prior to sampling, and was covered by plastic waterproof material secured by an elastic band.

Sampling occurred over a single season; the time required in surveying ground arthropods would not permit sampling in a second season. Spring was selected because this is the time of year when spiders, and other arthropods, are most active (Koch and Majer 1980). By sampling in only one season, an

underestimate of the true diversity of ground spiders is expected. Abbott ***et al.*** (2002) found 60 species in their first sampling season in Spring (1994), which increased to 385 species after the next nine sampling seasons.

Pitfall traps were placed in the ground on the 7-8 August (2001). One week later (15 August), water proof sleeves were removed and a plastic cup placed within the pitfall trap. As spiders are capable of climbing in and out of pitfall traps (Topping & Luff 1995), approximately 40mls of ethylene glycol was poured into each cup (Coyle 1981).

On the 30 August all samples were collected, arthropods were removed and placed into 70% his-ethanol. Arthropods were sorted to ordinal level using Harvey & Yen (1989) and Zborowski & Storey (1998). Spiders were identified to morpho-species (or 'recognisable taxonomic units' ***sensu*** Oliver & Beattie 1993) by Nadine Guthre (CALM). A morpho-species approach was used as < 20 % of spiders in Australia are named (Raven 1988 in Main 2001; Brennan ***et al.*** 2003).

6.2.3 Statistics

The statistical approach used in this chapter follows that described in Section 5.2.3.

6.3 RESULTS

6.3.1 General Description

Sixty species of ground-dwelling spiders were recorded from 22 families at Mt Dale (Appendix 4). Of these, 27 species were hunting species and 22 species were web builders. Eleven species were neither web-weaving nor ground hunting, and were not included in the analysis (Appendix 4). There was substantial variation in species richness among sites (range = 3 – 15 spp./site, $\bar{x}$ = 9.5 SE ± 0.6, spp./site). Some of this variation could be attributed to recent logging and fire disturbance (Table 6.1). Pairwise analysis shows that both burnt treatments ('burnt 2000' and 'gap logged/ burnt 2000') had fewer species that the three treatments that had not been burnt for five years ('undisturbed', 'gap logged 1990' and 'selectively logged < 1950') (Fig. 6.2). There was no difference in the number of species within forest that was gap logged and burnt in 2000 and unlogged forest burnt in 2000. Similarly, there were no differences in the number of species between sites in undisturbed forest, forest gap logged in 1990 and forest selectively logged prior to 1950 (Fig. 6.2).

Table 6.1 One-way ANOVA and power analysis testing whether there are differences in the species richness of spiders between treatments representing different disturbance regimes.

	Sum of Squares	**Degrees of Freedom**	**Mean Square**	**F-ratio**	**P value**	**POWER**
Between Groups	1.45	4	0.36	5.18	0.06	0.89
Within groups	1.05	15	0.07			
Total	2.50	19				

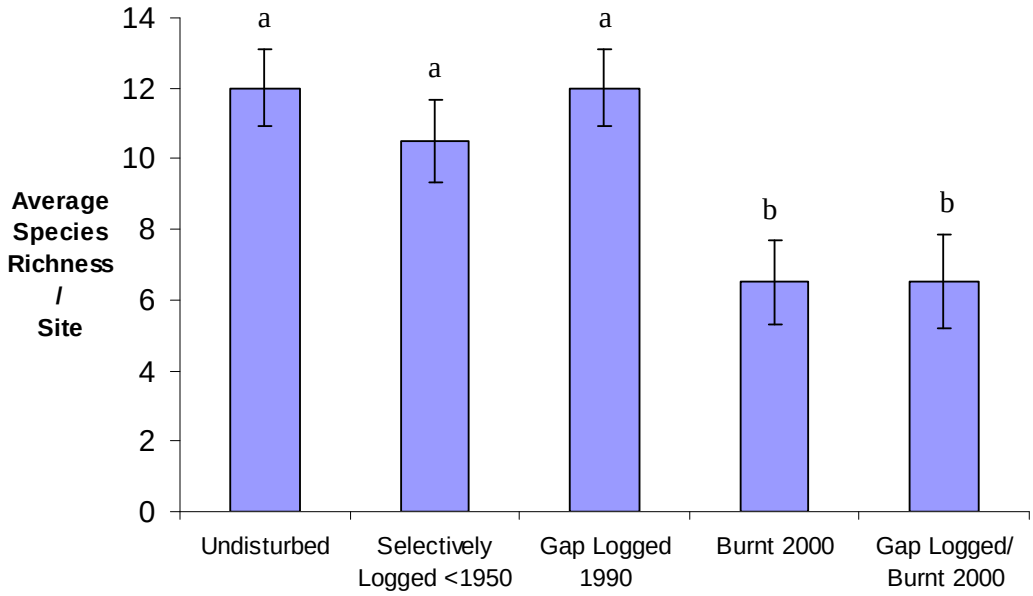


Fig. 6.2 The average (± SE) species richness of spiders per site for five treatments representing different disturbance regimes. Significant differences between sites were determined by one-way ANOVA. Different letters indicate significant differences ($p < 0.05$).

Multivariate analysis showed that there were different communities of ground spiders in different treatments (Global $R = 0.17$, $p < 0.05$). Pairwise analysis showed that the composition of spiders in forest that was gap logged in 1990 was different to the composition of spiders in undisturbed forest (Table 6.2). Pairwise analysis also showed communities of spiders in undisturbed forest were most similar to forest selectively logged prior to 1950. There was greater similarity in the assemblages of species in the two treatments disturbed by fire in 2000 ('gap logged/burnt 2000' and 'burnt 2000') and the two treatments disturbed by gap logging ('gap logged/burnt 2000' and 'gap logged in 1990') than among other treatments (Table 6.2).

Table 6.2 Summary of ANOSIM pairwise analysis testing whether the composition of spiders were different between five treatments representing different disturbance regimes. Highlighted rows denote significant differences between treatments ($p < 0.05$).

Groups	Global *R*	Statistic %
Undisturbed, Burnt 2000	0.25	8.6
Undisturbed, Gap Logged/ Burnt 2000	0.44	5.7
Undisturbed, Gap Logged 1990	0.41	2.9
Undisturbed, Selectively Logged <1950	-0.12	71.4
Burnt 2000, Gap Logged/ Burnt 2000	0.01	45.7
Burnt 2000, Gap Logged 1990	0.21	8.6
Burnt 2000, Selectively Logged <1950	0.25	8.6
Gap Logged/ Burnt 2000, Gap Logged 1990	0.04	37.1
Gap Logged/ Burnt 2000, Selectively Logged <1950	0.35	8.6
Gap Logged 1990, Selectively Logged <1950	0.15	25.7

There were differences in the composition of spiders in the two treatments burnt in 2000 compared to the three treatments burnt prior to 1995 on the ordination (Fig. 6.3). One site (Undisturbed '1') of the unburnt treatments is the only outlier in a pattern showing sites burnt in 2000 in one half of the plot and sites that had not been burnt for at least five years in the other half of the plot (Fig. 6.3). This pattern could be caused by the lower number of species recorded in sites burnt in 2000 compared to other treatments (Fig. 6.2). The ordination also indicates that sites in undisturbed forest were more dissimilar to sites in forest gap logged in 1990 than sites that had been selectively logged prior to 1950 (Fig. 6.3).

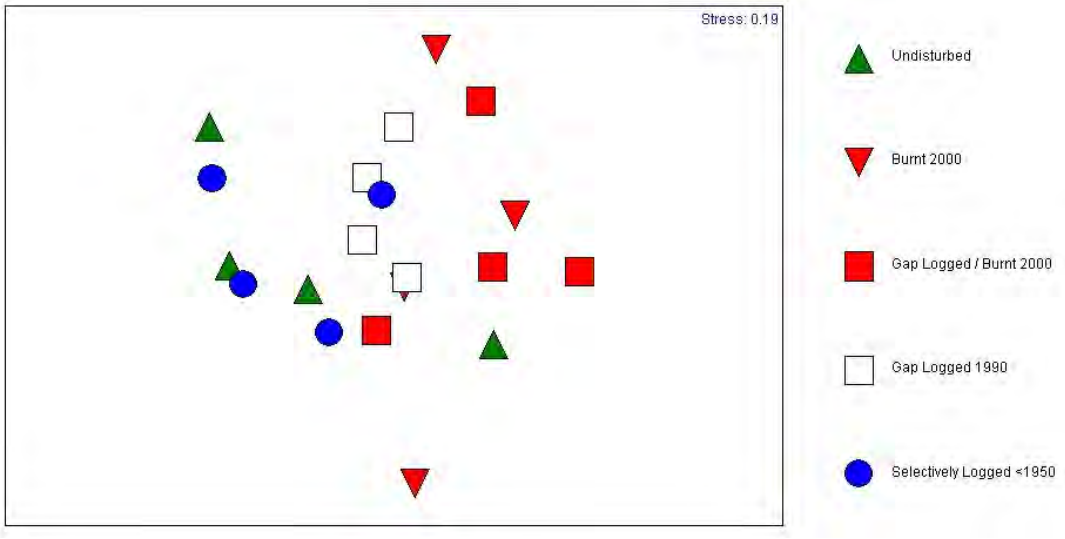


Fig. 6.3 Non-metric multidimensional scaling plot showing the effect of five disturbance regimes on the composition of spiders.

When sites are pooled within forest of similar disturbance regime, the three unburnt treatments had more species that both treatments burnt in 2000 (Fig. 6.4). Undisturbed forest contained more species than other treatments (Fig. 6.4). Forest selectively logged prior to 1950 had more species than forest gap logged in 1990 (Fig. 6.4). Forest that had been gap logged prior to regeneration burn had the lowest total number of species (Fig. 6.4).

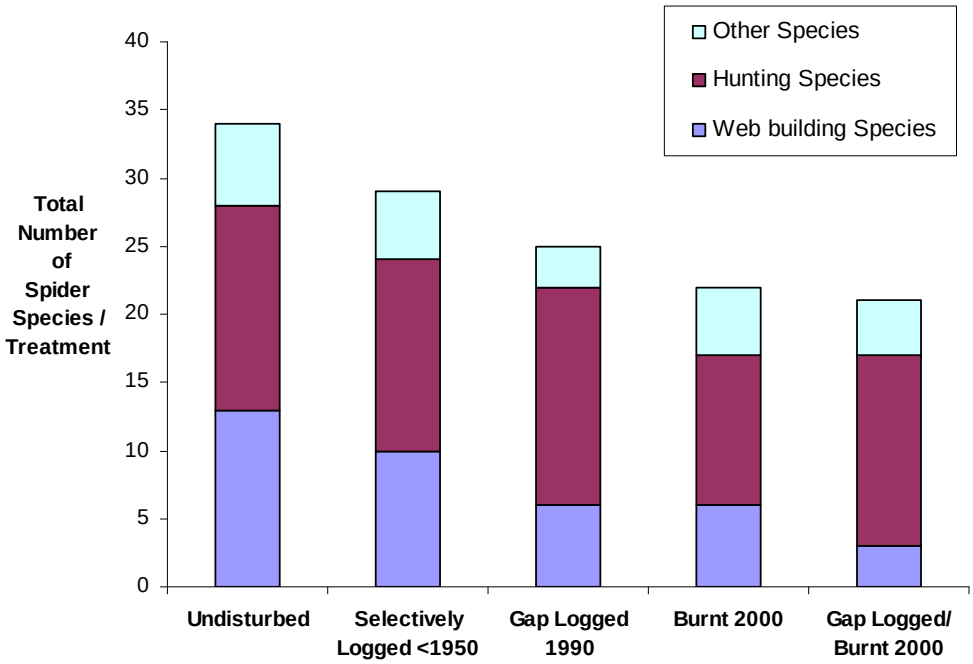


Fig. 6.4 The total number of spider species recorded within each treatment. Each treatment consisted of four sites. Spiders are categorised as hunting, web-building and other spider species.

6.3.2 Hunting Spiders

Species Richness

There was considerable variation in the number of hunting species among sites (range = 2 – 12 spp./site, $\bar{x}$ = 6.8 SE ± 0.6 spp./site). *Post hoc* analysis reveals eight sites per treatment would have provided the power to find a significant effect (estimated power = 0.98). When α was adjusted to account for the low statistical power observed in the ANOVA (power < 0.8), there were significant differences in average species richness between treatments ($p < 0.1$; Table 6.3). Pairwise analysis showed forest gap logged in 1990 had more species than all other treatments ($p < 0.1$; Fig. 6.5). Furthermore, both treatments burnt in 2000 had fewer species than the three treatments that were not burnt in 2000 ($p < 0.1$; Fig. 6.5). Forest selectively logged prior to 1950 had a similar number of species as undisturbed forest (Fig. 6.5). Similarly, the species richness of spiders in both treatments burnt in 2000 was equivalent.

Table 6.3 One-way ANOVA and power analysis testing whether there are differences in the species richness of hunting spiders between treatments representing different disturbance regimes.

	Sum of Squares	Degrees of Freedom	Mean Square	F-ratio	P Value	POWER
Between Groups	1.62	4	0.40	2.77	0.06	0.65
Within groups	2.19	15	0.15			
Total	3.80	19				

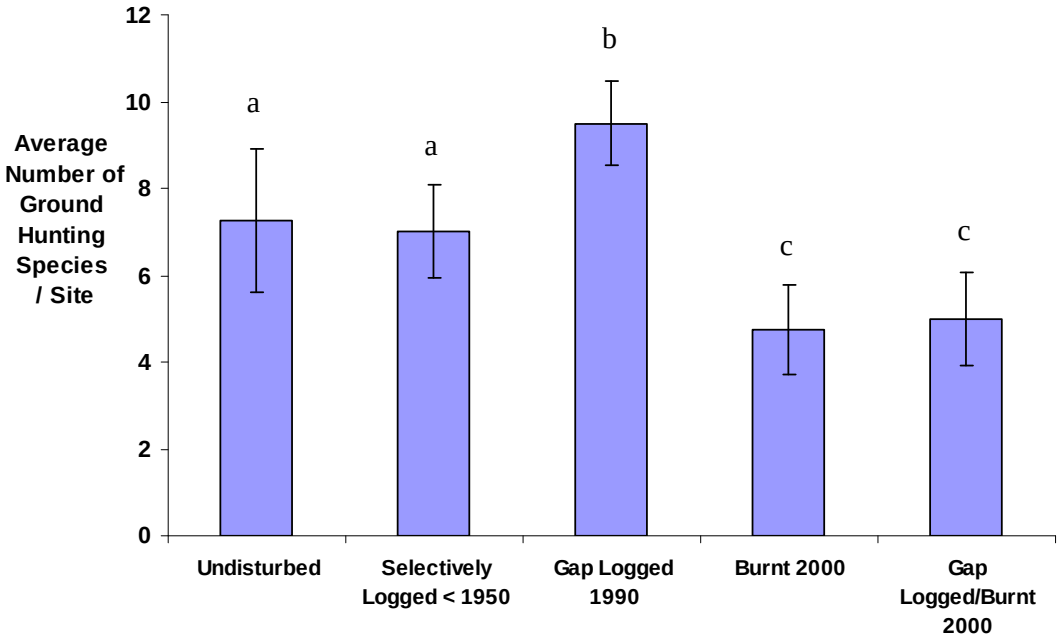


Fig. 6.5 The average (± SE) species richness of hunting spiders per site for five treatments representing different disturbance regimes. Significant differences between sites were determined by one-way ANOVA. Different letters indicate significant differences ($p < 0.1$).

Community Composition

There was no significant difference in the composition of hunting species between treatments (Global $R = 0.46$, $p > 0.05$; Fig. 6.6). Seven hunting species (26 % of all hunting species) were recorded across all treatments, where as 16 (59 %) species were recorded at less than four sites (Table 6.4). These 'uncommon' species were distributed evenly across treatments ('undisturbed' = 6 spp., 'selectively logged < 1950' = 4 spp., 'gap logged 1990' =5 spp., 'gap logged/ burnt 2000' = 5 spp. and 'burnt 2000' = 4 spp.). There was overlap in the composition of ground hunting spiders among the five treatments on the

ordination (Fig. 6.6), possibly resulting from the relatively even number of uncommon species among the five treatments. This ordination also indicated that the composition of species between sites burnt in 2000 ('burnt 2000' and 'gap logged/ burnt 2000' treatments) was more variable than sites that had not been burnt for five years ('undisturbed', 'gap logged 1990' and 'selectively logged < 1950'). Sites in forest selectively logged prior to 1950 also appeared to have greater similarity in species assemblages with sites that had been gap logged in 1990 than sites other treatments (Fig. 6.6).

Table 6.4 The hunting spiders recorded in five treatments representing different disturbance regimes ('Un' – Undisturbed, 'SL 50' – Selectively Logged < 1950, 'GL 90' – Gap Logged 1990, 'GLB' – Gap Logged/ Burnt 2000, 'B' – Burnt 2000). Numbers represent the number of sites that species were recorded.

FAMILY	SPECIES	Un	SL 50	Gl 90	GLB	B
GNAPHOSIDAE	Gnaphosidae sp. 1	4	4	4	3	3
LIOCRANIDAE	Licranidae sp.1	1	0	0	1	1
MITURGIDAE	Miturgidae sp. 1	0	0	0	0	1
	Miturgidae sp. 2	2	2	2	1	1
	Miturgidae sp. 3	0	0	0	1	0
	Miturgidae sp. 4	0	0	0	1	0
	Miturgidae sp. 5	1	0	1	0	0
	Miturgidae sp. 6	0	1	2	0	0
	Miturgidae sp. 7	1	0	1	1	0
	Miturgidae sp. 8	1	0	2	0	2
	Miturgidae sp. 9	0	1	0	0	0
OONOPIDAE	*Gamasomorpha* sp. 1	0	1	4	1	0
	Myrmopopaea sp. 1	3	3	4	3	2
	Opopaea sp. 1	0	0	1	0	0
	Orchestina sp. 1	0	0	1	0	0
SALTICIDAE	*Lycidas michaelseni*	4	3	2	1	2
	Lycidas sp. 1	1	3	3	0	0
	Lycidas sp. 2	3	2	3	1	1
	Lycidas chysomelas	0	0	0	1	0
	Maratus mungiach	2	2	4	3	4
	Hypoblemum sp. 1	0	1	0	0	0
	Salticidae Genus 1 sp. 1	3	4	3	1	2
	Salticidae Genus 1 sp. 2	1	1	0	0	0
	Salticidae Genus 2 sp. 1	1	1	1	1	0
	Salticidae Genus 3 sp. 1	1	0	0	0	0
SPARASSIDAE	Sparassidae sp. 1	0	0	0	0	1
Total Number of Species / Treatment		15	14	16	14	11

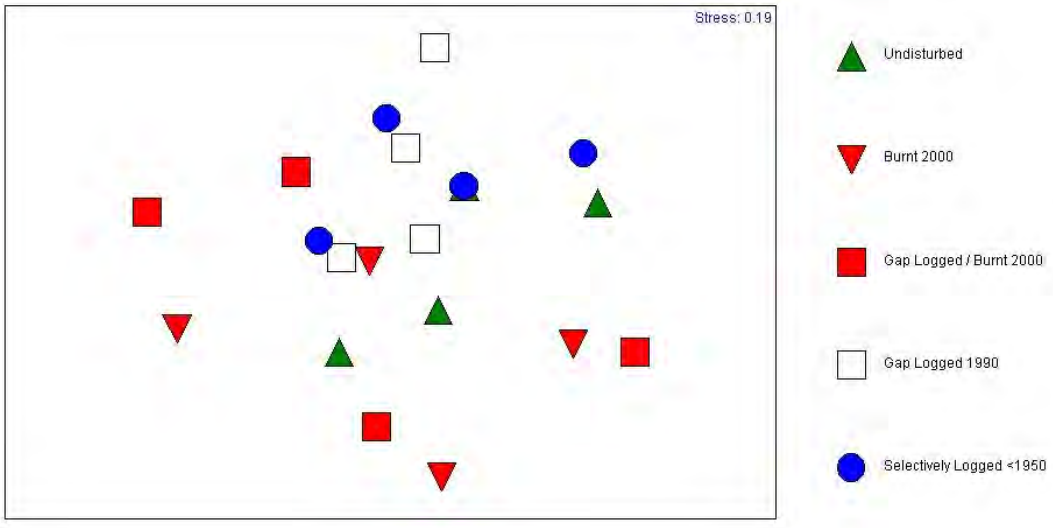


Fig. 6.6 Non-metric multidimensional scaling plot showing the effect of five disturbance regimes on the composition of hunting spiders.

Two species (*Gamasomorpha* sp. 1 and *Maratus mungiach*) contributed to there being more species in gap-logged 1990 forest than unlogged forest (Fig. 6.5). *Gamasomorpha* sp. 1 was not found in undisturbed forest, but occurred in all four sites that were gap logged in 1990 (Table 6.4) *Maratus mungiach* was also more commonly recorded in forest gap logged in 1990 than in undisturbed and forest selectively logged prior to 1950 (Table. 6.4). Three species (*Lycidas* sp. 1, *Lycidas* sp. *2* and Salticidae Genus 1 Sp. 1) were recorded less frequently in forest burnt in 2000 than in forest burnt prior to 1995.

6.3.3 Web-Builders

Species Richness

The number of web-building species was low for all sites, averaging 2.8 species / site (SE ± 0.32, range 1-5). Despite this low species richness, there were significantly different numbers of web-building species between treatments at standard levels of significance ($p < 0.05$; Table 6.5). Pairwise analysis showed that more species of web building spiders were recorded in unlogged forest and

forest selectively logged prior to 1950 than in both burnt treatments ($p < 0.05$; Fig. 6.7). There no difference in the number web building spiders in sites gap logged in 1990 and the two treatments burnt in 2000 ($p > 0.05$; Fig. 6.7). There were also no differences in the number of species in undisturbed forest, forest selectively logged prior to 1950 and forest gap logged in 1990 ($p > 0.05$; Fig 6.7).

Table 6.5 One-way ANOVA and power analysis testing whether there are differences in the species richness of web-building spiders between treatments representing different disturbance regimes.

	Sum of Squares	Degrees of Freedom	Mean Square	F-ratio	P value	POWER
Between Groups	3.83	4	0.96	5.06	.01	0.88
Within groups	2.84	15	0.19			
Total	6.68	19				

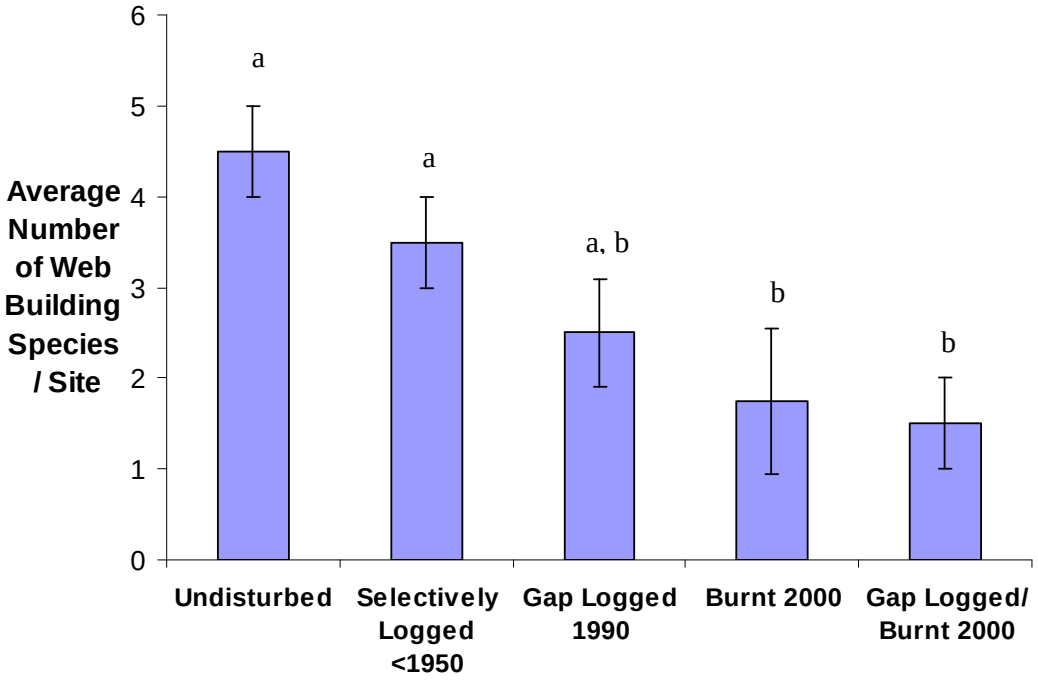


Fig. 6.7 The average (± SE) species richness of web-building spiders per site for five treatments representing different disturbance regimes. Significant differences between sites were determined by oneway ANOVA. Different letters indicate significant differences ($p < 0.05$).

Species Composition

Sixteen species (72 % of all web building species) were recorded at less than four sites (Table 6.6). This created significant variation in the composition of web

building species between each site. More 'uncommon' species (i.e., those species recorded at less than four sites) were recorded in unlogged sites ('undisturbed' = 10 spp. and 'burnt 2000' = 5 spp) and forest selectively logged prior to 1950 (n = 6 spp.) than in forest gap logged in 1990 (n = 3 spp.) and gap logged/ burnt in 2000 (n = 1 spp.). Within this variation, significant differences in the composition of web-building species between treatments were found (Global $R = 0.305$; $p < 0.05$). Pairwise analysis showed that sites in undisturbed forest contained a different composition of web-building species than all treatments except forest selectively logged prior to 1950 (Table 6.7). Moreover, the composition of species in forest selectively logged prior to 1950 was different to the community found in forest gap logged in 1990 (Table 6.7). The composition of species in undisturbed forest was most similar to forest selectively logged prior to 1950 (Table 6.7). The two treatments that were burnt in 2000 also had more species in common than with the three treatments that had not been burnt since 1995 (Table 6.7). Ordination were not possible because the stress was greater than 0.2 for both two and three dimension ordinations.

Table 6.6 The web-building spiders recorded in five treatments representing different disturbance regimes ('Un' – Undisturbed, 'SL 50' – Selectively Logged < 1950, 'GL 90' – Gap Logged 1990, 'GLB' – Gap Logged/ Burnt 2000, 'B' – Burnt 2000). Numbers represent the number of sites that species were recorded.

FAMILY	SPECIES	Un	SL 50	Gl 90	GLB	B
AMAUROBIIDAE	Amaurobiidae sp. 1	1	0	0	0	1
	Amaurobiidae sp. 2	0	1	0	0	1
ARANEIDAE	Araneidae sp. 1	1	1	0	0	0
	Araneidae sp. 2	2	0	0	0	0
CYATHOLIPIDAE	*Matilda* sp. 1	0	0	0	0	1
DESIDAE	Desidae sp. 1	1	0	0	0	0
LINYPHIIDAE	Linyphiidae- sp. 1	2	0	0	0	0
	Linyphiidae- sp. 2	2	0	0	0	0
MICROPHOLCOMMATIDAE	Textricella sp. 1	0	0	1	0	0
	Micropholcommatinae sp. 1	2	2	0	0	0
	Micropholcommatinae sp. 2	2	2	0	0	0
STIPHIDIIDAE	Stiphidiidae sp. 1	1	0	1	0	0
THERIDIIDAE	*Dipoena* sp.1	1	1	0	1	0
	Dipoena sp. 2	1	0	0	0	0
	Gmogola sp.1	0	1	1	0	0
	Hadrotarsus sp. 1	0	2	0	0	0
	Hadrotarsus sp. 2	1	2	1	0	0
	Hadrotarsus sp. 3	0	0	0	0	1
	Phoroncidia sp.1	0	1	4	4	2
	Theridiidae sp. 1	0	1	2	0	0
	Theridiidae sp. 2	0	0	0	0	1
	Theridiidae sp. 3	1	0	0	1	0
AMAUROBIIDAE	Amaurobiidae sp. 1	1	0	0	0	1
TOTAL NUMBER OF SPECIES / TREATMENT		**13**	**10**	**6**	**3**	**6**

Table 6.7 Summary of ANOSIM pairwise analysis testing whether the composition of web-building spiders were different between five treatments representing different disturbance regimes. Highlighted rows denote significant differences between treatments ($p < 0.05$).

Groups	Global *R*	Statistic %
Undisturbed, Burnt 2000	0.42	2.9
Undisturbed, Gap Logged/ Burnt 2000	0.80	2.9
Undisturbed, Gap Logged 1990	0.83	2.9
Undisturbed, Selectively Logged <1950	0.08	40.
Burnt 2000, Gap Logged/ Burnt 2000	0.08	34.3
Burnt 2000, Gap Logged 1990	0.06	25.7
Burnt 2000, Selectively Logged <1950	0.14	22.9
Gap Logged/ Burnt 2000, Gap Logged 1990	0.16	14.3
Gap Logged/ Burnt 2000, Selectively Logged <1950	0.55	2.9
Gap Logged 1990, Selectively Logged <1950	0.34	14.3

The high number of species that were seldom recorded (less than four sites) meant that identifying species that contributed to differences consisted on relying on relatively weak trends. For example, five species (Linyphiidae sp. 1., Linyphiidae sp. 2, Micropholcommatinae sp. 1, Micropholcommatinae sp. 2, Araneidae sp. 2) of web-building spider were recorded in at least two undisturbed sites but not recorded in any recently disturbed sites (Table 6.6;

Table 6.8). Conversely, one species (*Phoroncidia* sp. 1) was more commonly recorded in both burnt treatments and forest gap logged in 1990 forest but was not recorded in undisturbed forest (Table 6.6; Table 6.8). *Theridae* sp. 1 was also recorded in two sites that were gap logged in 1990 but not in undisturbed forest. Small trends also suggest that forest selectively logged prior to 1950 contained a more diverse community than forest gap logged and burnt in 2000 (Table 6.6; Table 6.8). Four species (Micropholcommatinae sp. 1, Micropholcommatinae sp. 2, *Hadrotarsus* sp. 1 and *Hadrotarsus* sp. 2) were recorded in forest selectively logged prior to 1950 but not in recently gap logged/burnt forest (Table 6.6; Table 6.8). One species (*Phoroncidia* sp. 1) was more commonly recorded in recently gap logged forest than forest selectively logged prior to 1950 (Table 6.6; Table 6.8).

Table 6.9 The web-weaving species that contributed to differences between paired treatments identified as containing different assemblages of species by ANOSIM analysis (Table 6.7). Species are grouped within the 'treatment' in which they were frequently encountered.

Treatments	**Species Contributing to Difference Between Treatments** (% Contribution to Dissimilarity)
Undisturbed	Linyphiidae sp. 1 (9.34)
	Micropholcommatinae sp. 1 (9.34)
	Micropholcommatinae sp. 2 (9.34)
	Araneidae sp. 2 (7.65)
	Linyphiidae sp. 2 (7.65)
Burnt 2000	*Phoroncidia* sp. 1 (8.77)
Undisturbed	Linyphiidae sp. 1 (9.49)
	Micropholcommatinae sp. 1 (9.49)
	Micropholcommatinae sp. 2 (9.49)
	Araneidae sp. 2 (9.49)
	Linyphiidae sp. 2 (7.74)
Gap Logged / Burnt 2000	*Phoroncidia* sp.1 (17.23)
Undisturbed	Linyphiidae sp. 1 (8.16)
	Micropholcommatinae sp. 1 (8.16)
	Micropholcommatinae sp. 2 (8.16)
	Araneidae sp. 2 (6.82)
	Linyphiidae sp. 2 (6.82)
Gap Logged 1990	*Phoroncidia* sp.1 (14.98)
	Theridiidae sp. 1 (7.13)
Selectively logged < 1950	Micropholcommatinae sp. 1 (9.49)
	Micropholcommatinae sp. 2 (9.49)
	Hadrotarsus sp. 1 (9.49)
	Hadrotarsus sp. 2 (9.49)
Gap logged / Burnt 2000	*Phoroncidia* sp.1 (15.12)

6.4 Discussion

Logging is not analogous to fire with respect to the way the spider fauna responds to these disturbances. There were different assemblages of spiders in forest disturbed by gap logging in 1990 than in undisturbed forest. This ten year time period is a longer period of time than fire affects spider communities, with other studies in jarrah forest showing the spider fauna recovering six years after fire (Brennan 2002). It is not known how long gap logging affects spider communities. Forest selectively logged prior to 1950 contained similar diversity (both species richness and composition) as undisturbed forest, indicating that spider communities could take less than 50 years to recover from gap logging disturbance. However, gap logging is more intensive than selective logging (Chapter 4), suggesting spider communities may take longer to recover from gap logging than selective logging. Differences between the effects of gap logging and fire disturbance highlight the need to maintain viable populations of spiders in unlogged forest and forest that has recovered from logging disturbance.

6.4.1 The Impact of Fire in Unlogged and Logged Forest

The diversity of spiders in forest burnt in 2000 was lower than in forest last burnt before 1995. This was expected, as fire can eliminate populations of spiders along with approximately 90% of the micro-arthropod food base (Huhta 1971; McIver *et al.* 1992). In jarrah forest, studies have shown it takes at least 13 months for arthropods to recover to pre-fire levels of abundance after an autumn burn (Majer 1984); and more than three years for spider communities to recover (Brennan 2002). A loss of species is thought to occur because fire can reduce the resources available to spiders (Huhta 1971; Buddle *et al.* 2000). Spider

communities react to habitat variables that are eliminated by fire, including leaf litter depth, cover, and vegetation cover (both dead and alive) (Brennan 2002).

Spider communities recover after fire either by having individuals survive the passage of fire (Main 1976; Whelan ***et al.*** 1980), or by juveniles and adults quickly recolonising the area (Main 1976; Main 1995). In my research, species from 10 families were found in recently burnt, but unlogged, jarrah forest (AMAUROBIIDAE, CYATHOLIPIDAE, GNAPHOSIDAE, LIOCRANIDAE, MITURGIDAE, OONOPIDAE, SALTICIDAE, SPARASSIDAE, and THERIDIIDAE). Other studies have shown that some species are able to survive fire by sheltering among leaf bases of grass trees or under thick bark (e.g., members of the SALTICIDAE and SPARASSIDAE family) (Whelan ***et al.*** 1980; Brennan 2002). Other species recolonise after fire by ambulatory means (e.g., members of GNAPHOSIDAE, OONOPIDAE and SALTICIDAE) (Main 1976; Brennan 2002). At Mt Dale, three species (*Lycidas* sp. 1, *Lycidas* sp. *2* and Salticidae Genus 1 Sp. 1) were regularly record in forest that had not been burnt since 1995 but were not recorded in forest burnt in 2000, indicating that they had not recovered from fire disturbance.

Arachnologists have argued that spider families that evolved when the Australian landmass was still part of Gondwana (Gondwana relicts) are sensitive to the increasing regularity of fire as a result of human practices (Main 2001). This is because these relictual taxa are now restricted to mesophytic (moist) habitats either in rainforest or wet sclerophyll forest, or microhabitats that retain characteristics of Gondwanan habitats (e.g., caves) (Main 1998 in Main 2001). Although the northern jarrah forest is a relatively dry forest that has been

frequently disturbed by fire for thousands of years, several relictual families occur in this region (Main 2001). At Mt Dale, two families were recorded that are considered Gondwanan relicts (CYATHOLIPIDAE and MICROPHOLCOMMATIDAE) (Main 2001). The three Micropholcommatidae species were recorded in unburnt forest. Two of the species (Micropholcommatinae sp. 1 and Micropholcommatinae sp. 2) were only found in unlogged forest and forest that had been selectively logged prior to 1950, whereas the third species (Textricella sp. 1) was only recorded once in forest gap logged in 1990. In contrast, the one record of a Cyatholipidae species was in unlogged forest that had been recently burnt, indicating this species either survived the fire, or quickly re-colonised following fire.

There were differences in the composition of spiders between unlogged/ burnt forest and gap logged/burnt forest. It is proposed that gap logging prior to fire causes the reduction of web building spiders as only one web-weaving species was recorded in gap logged sites as opposed to five species recorded in unlogged sites. As outlined in Section 5.4.1, these differences could result from disturbance of gap logging prior to the fire event, or alternatively could be a result of different fire intensities. With respect to the first explanation, previous research has revealed that gap logging without the additional impact of post logging fire can affect the abundance of some arthropods groups (Strehlow ***et al.*** 2002.) Studies in other temperate forests have shown that intensive logging can affect the web building spiders because particular microhabitats are disturbed (see Coyle 1981; McIver ***et al.*** 1992; Buddle ***et al.*** 2000; Willet 2001).

The alternative explanation for differences between the two burnt treatments is that fire could be more intense in gap logged sites due to enhanced fuel loads

compared with that in unlogged sites. Sites exposed to high intensity fires have been shown to have fewer species of arthropods that sites exposed to medium intensity burns (Van Huerck *et al.* 1998). Fires that have higher intensities could have significant impact on communities because they remove more of the sheltering sites (e.g., under logs, in balga) utilised by many spider species in order to survive. Given the unique habitat requirements of different spider species in jarrah forest (Main 2001; Brennan 2002), it is likely that a combination of factors cause the differences in the spider fauna observed between gap logged/burnt forest and unlogged/burnt forest.

6.4.2 Spider Diversity 10 years after Gap Logging

The spider fauna in forest gap logged in 1990 was different from the fauna found in unlogged forest. Although the number of species is almost equivalent between gap logged and undisturbed forest, species assemblages were different. The effect of gap logging appears to be similar to the effects caused by clearfelling in many other forests, where some mature forest 'specialists' (web-builders) are replaced by 'eurocryptic' species (hunters) (Coyle 1981; Heliovaara 1984; McIver *et al.* 1992; Pajunen *et al.* 1995). In my research, five species of web building spider (Linyphiidae sp. 1., Linyphiidae sp. 2, Micropholcommatinae sp. 1, Micropholcommatinae sp. 2, Araneidae sp. 2) were only recorded in undisturbed forest, where as two hunting species (*Gamasomorpha* sp. 1 and *Maratus mungiach*) were more commonly recorded in forest gap logged in 1990. Changes in the guild structure of spider communities following logging are thought to be a consequence of many interrelated factors, including changes in solar radiation, moisture, spatial structure (to allow for the frameworks to make

webs), prey-availability, predation, competition and capacity for dispersal (Huhta 1971; McIver *et al.* 1992).

This research shows that previously identified habitat variables that were important in determining spider species taxon richness after fire (leaf litter cover, and vegetation cover) (see Brennan 2002) are not major determinants of spider diversity after gap logging. This is because these attributes were equivalent between forest gap logged in 1990 and unlogged forest (see Chapter 4; also noted by Van Huerck *et al.* 1998). It is not in the scope of this research to tease apart the factors likely to influence the spider fauna in jarrah forest following logging. To do this would require manipulative experiments (e.g., removing competitors, changing prey availability, changing moisture regimes) as well detailed assessment of microhabitats required by each species of spider. However, intensive logging in other temperate forests does affect spider communities for at least thirty years (McIver *et al.* 1990; McIver *et al.* 1992; Buddle *et al.* 2000). Until research indicates otherwise, forest managers must recognise that gap logging could have ongoing effects on the composition of ground spiders.

6.4.3 Spider Diversity 50 years after Selective Logging

Forest selectively logged prior to 1950 has significant conservation value for spiders. Several species that appear to be affected by gap logging are found in these forests (e.g., Micropholcommatinae sp. 1, Micropholcommatinae sp. 2 and *Hadrotarsus* sp. 2). Some species of the MICROPHOLCOMMATINAE family are Gondwanan relicts and restricted to mesophytic habitats in forests (Main 2001). Given the similarities in ground attributes between forest selectively logged prior to 1950 and undisturbed forest (Chapter 4), this research suggests that forest

selectively logged before 1950 could be used to conserve spider communities that are affected by contemporary logging practices.

Although forest selectively logged prior to 1950 had statistically similar levels of α and β diversity of both guilds, there were small differences in diversity between these treatments. When sites were pooled, more species of spider were found in undisturbed forest than forest selectively before 1950. This occurred because more species of uncommonly recorded web builders (i.e., those species found at less than four sites) were found in unlogged sites than in forest that had been disturbed by logging. For example, small web-weaving species from five families (LINYPHIIDAE, ARANEIDAE, STIPHIDIIDAE, DESIDAE, AMAUROBIIDAE) were only recorded in unlogged forest that had not been burnt for five years. This difference means that a landscape of unlogged forest would have more species than a mosaic of forest selectively logged prior to 1950 and unlogged forest. This research therefore shows that protecting unlogged forest will conserve more species than protecting equivalent areas in production forest, and will also provide habitat for species of spider that are deleteriously affected by past and present logging practices.

6.4.4 Future Research

Research must continue investigating how contemporary logging practices impact on spider communities. My research shows that spider communities are affected by gap logging for a minimum of 10 years, and the impact could exist for a longer period of time. Given that this contemporary logging is likely to occur throughout large areas of the jarrah forest (Burrows *et al.* 2002b), future research needs to assess when spider communities return to pre-logging levels.

Moreover, manipulative experiments need to be undertaken to assess what habitat attributes need to be protected in order to successfully conserve communities in recently disturbed sites.

Future research should not necessarily focus on assessing the entire spider community. My research found that web-building spiders (especially LINYPHIIDAE, ARANEIDAE, STIPHIDIIDAE, DESIDAE, AMAUROBIIDAE) could be a focus of future research. This is because these taxa appear to be negatively affected by contemporary logging practices. Other studies have shown that ground or burrow-dwelling taxa (e.g., many MYGALOMORPHAE species) are particularly vulnerable to burning (Main 1995) and therefore also should be targeted because they could be at risk from contemporary practices that incorporate post logging burning practices.

Future research also needs to recognise that methods in addition to pitfall trapping need to be employed in order to sample the entire spider fauna in jarrah forest. Although pitfall traps are internationally accepted as a way of sampling spider communities, these traps only sample surface-active taxa. Other methods, including intensive area searches within specified micro habitats and vacuum sampling (see Brennan 2002), needs to occur to supplement pitfall trapping.

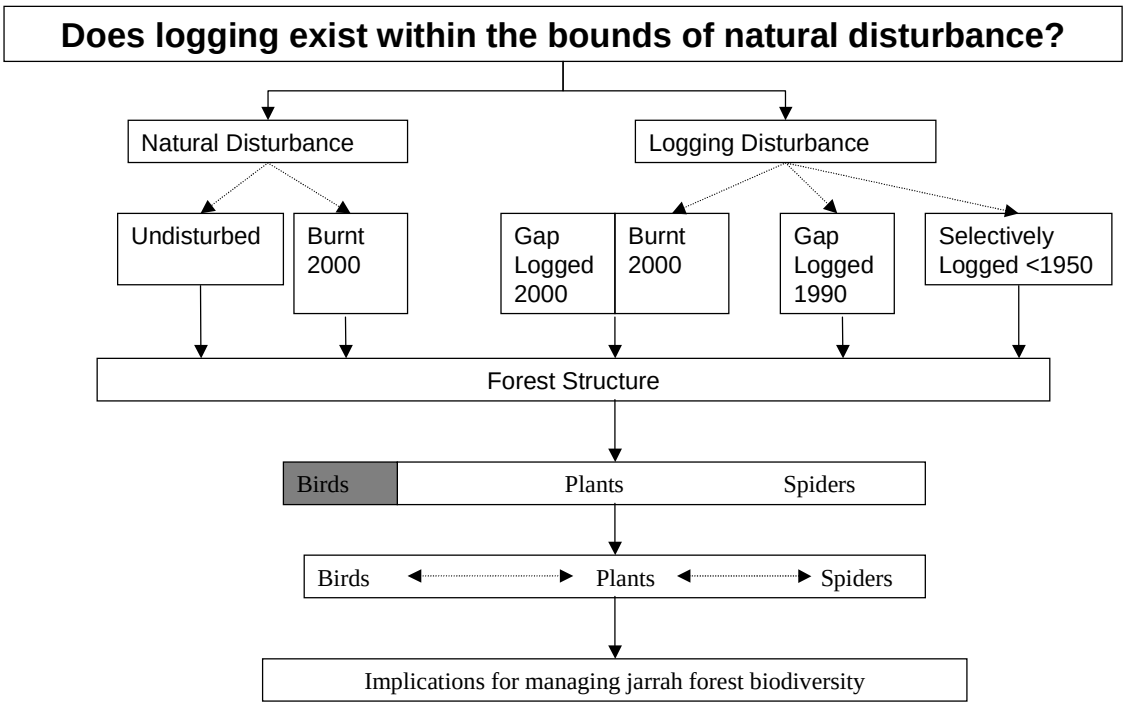
6.5.5 Conclusion

This research does not support findings by Strehlow *et al.* 2002 and Abbott *et al.* 2002 that assert that the impact of logging was of only short duration, and that ground arthropod communities are resilient to logging because of the climatic

and fire history of Western Australia. Instead, it shows that species that require specific attributes (e.g., web-builders) have a low degree of tolerance to severe disturbance such as gap logging and fire. The problems of determining the true diversity of invertebrates in any ecosystem, and of documenting the interactions between species or functional guilds, are formidable and are not likely to be resolved for a very long time (New 1987). Future studies should assess whether particular species (particularly small web-building species) are re-establishing in disturbed forests rather than simply focusing on total species numbers. This research provides insights that have not been previously documented: that unlogged and forest recovering from early selective logging is needed to conserve elements of ground spider diversity, and that contemporary silvicultural practices have ongoing effects on spider communities.

CHAPTER 7

Birds



Chapter 7

7.1 Introduction

Birds have been extensively surveyed in jarrah forest (reviewed by Abbott 1999). Extensive taxonomic knowledge, conspicuous behaviour and predictable responses to disturbance make birds useful for testing ecological theory and monitoring environmental change (Recher 1988; Temple & Wiens 1989; Hansen *et al.* 1993). Although the jarrah forest contains only about half the species present in similar sized, comparable areas of temperate forest in eastern Australia (Wykes 1985; Abbott 1999), studies have shown that the bird community in jarrah forest is useful for studying disturbance caused by such activities including mining (e.g., Collins *et al.* 1985; Curry & Nichols 1986), fire (e.g., Kimber 1974b; Davies 1979) and logging (e.g., Norwood *et al.* 1995; Craig 1999). Therefore, birds were selected to investigate the effects of fire and logging at Mt Dale.

The impact fire has had on bird diversity has been extensively studied. The most recent review showed that the effect fires have on bird diversity is difficult to predict (Burbidge 2003). Fire frequency, intensity, the season and the extent of fire; and climatic variation before, and after, a given fire, will all influence how birds respond to fire (Burbidge 2003). This range of variation has led to calls for long-term, well replicated and controlled studies on the response of bird communities to different fire regimes so that scientists can more confidently predict and manage for fire disturbance (Christensen *et al.* 1985; Burbidge 2003).

Studies have generally shown that fire has an immediate and severe impact on bird species richness and composition. Some species including the Australian

Raven (*Corvus coronoides*) and the Australian Magpie (*Gymnorhina tibicen*) take advantage of temporarily increased availability of food (e.g., fire killed prey) (Kimber 1974b). In contrast, many species are negatively affected because much of their foraging and breeding habitat (including leaf litter, understorey vegetation and, after intense fires, canopy cover) is destroyed by fire (Christensen & Abbott 1989). With respect to breeding habitat, the immediate impact of fire is thought to depend on the season in which the fire occurred. Spring fires are hypothesised to be more deleterious than autumn burns because fire in spring jeopardizes breeding effort of many species (Davies 1979). Managers of the jarrah forest claim there is no empirical proof of this (Abbott 1999), although no study has ever convincingly shown that spring fires are equivalent to autumn fires with regard to the breeding success of birds in jarrah forest.

Reductions in foraging resources following fire are of short duration. As with other temperate forests (see review in Woinarksi & Recher 1997), there is an increase in arthropod abundance on the regenerating vegetation following fire in jarrah forest (Kimber 1974b). The number of bird species has been shown to increase above pre-fire levels for two years after low intensity fire (Christensen & Kimber 1975) and three years after high intensity fire (Christensen *et al.* 1985). In the study of Christensen *et al.* (1985), 13 species appeared after fire that had not been recorded before this disturbance. As such, disturbance by fire is thought to have a positive impact on bird diversity because it increases foraging resources available to many species (Abbott 1999).

There have been several studies that have assessed how birds are affected by logging. Abbott and van Huerck (1985a) assessed a silvicultural practice no

longer utilised (half the canopy being thinned, and all *Banksia* removed). This study found two species (Grey Fantail *Rhipidura fuliginosa* and Rufous Treecreeper *Climacteris rufa*) were recorded more frequently in logged areas, and no species were negatively affected by logging. Abbott & van Huerck (1985a) also showed that two species (Tree Martin *Hirundo nigricans* and Australian Ringneck *Barnardius zonarius*) became more abundant in adjacent yarri (*Eucalyptus patens* Benth.) dominated forest after the jarrah forest was logged.

In a more recent study, Norwood *et al.* (1995) investigated bird diversity and abundance in gap logged coupes, edges between gap logged forest and mature forest (forest that had been previously disturbed by selective logging), and mature forest. Five species were more frequently found in gaps (Red-capped Parrot *Platycercus spurius*, Tree Martin, Red-winged Fairy-wren *Malurus elegans*, Dusky Woodswallow *Artamus cyanopterus* and Grey-breasted White-eye *Zosterops lateralis*). In contrast, two species (Grey Fantail and Inland Thornbill *Acanthiza pusilla*) were more frequently recorded in mature forest than in gap logged forest. No species preferred edges. This study also showed that almost twice as many birds were detected in gap logged forest and in edges compared with those in mature forest. The lack of similarity in the results of Norwood *et al.* (1995) and Abbott & van Huerck (1985a) is likely to reflect the different types of logging assessed in each study, as well as the different methodologies used to assess bird diversity.

The most comprehensive assessment of the effects of logging on bird diversity has been conducted in the Kingston Study (see Section 2.6). This research

showed that contemporary logging has an immediate impact on the density and diversity of birds (Craig 1999). Of the 28 species with the sufficient abundance to allow for analysis, 10 changed in abundance for one year after gap and shelterwood logging. Four species increased in abundance after logging (Splendid Fairy-wren ***Malurus splendens***, Red-winged Fairy-wren, Dusky Woodswallow and White-browed Scrubwren (***Sericornis frontalis***) and six species decreased in abundance (Golden Whistler ***Pachycephala pectoralis***, White-naped Honeyeater ***Melithreptus chloropsis***, Spotted Pardalotte ***Pardalotus punctatus***, Striated Pardalotte ***Pardalotus striatus***, Grey Fantail and Western Gerygone ***Gerygone fusca***). There were no differences in population changes after gap logging or shelterwood logging.

A second component of Craig's (1999) research investigated how gap logging and shelterwood logging affected the breeding and foraging habitat for four species, the Western Yellow Robin (***Eopsaltria grisogularis***), Rufous Treecreeper, Golden Whistler and White-naped Honeyeater (Craig ***et al.*** 2001). The two species that foraged in the canopy (Golden Whistler and White-naped Honeyeater) declined after logging, whereas the two ground foraging species (Western Yellow Robin and Rufous Treecreeper) were not affected. Investigating how foraging guilds are affected by logging and fire therefore appears to be an appropriate way to assess how birds are affected by these disturbances.

In a continuation of the Kingston study, one species (Western Gerygone) was shown to be affected for at least 6 years (Abbott ***et al.*** 2003). Other species may have also been detrimentally affected but no definitive conclusions could be

made because of large inter-annual variation in populations (Abbott *et al.* 2003). Further research is therefore required to assess whether logging has ongoing affects on the diversity of jarrah forest birds. My research will add to the initial findings of the Kingston study by addressing the following three questions:

1) Are there differences in the richness or assemblage of bird species within different guilds in forest recently disturbed by fire compared with forest recently disturbed by gap logging and fire?

2) Does the richness or assemblage of bird species within different guilds recover 10 years after gap logging?

3) Does the richness or assemblage of bird species within different guilds recover 50 years after selective logging?

7.2 Methods

7.2.1 Selecting Guilds

Logging influences the foraging resources available to birds in many Australian forests (Recher 1991; Wardell-Johnson & Williams 2000; Williams *et al.* 2001; Kavanagh & Stanton 2003). In jarrah forest, foraging guilds have been successfully used to investigate the short-term impacts of fire (Kimber 1974a) and logging (Craig 2002). In following this approach, species recorded at Mt Dale were placed into one of five guilds, each representing the routine manner and location the birds gather food (***sensu*** Mac Nally 1994). The foraging descriptions of bird species in jarrah forest by Wykes (1985) and Abbott (1999) were used to categorise species as:

1) *Tree Foraging*: species that predominantly forage in the canopy of trees,
2) *Ground/Shrub Foraging*: species that predominantly forage on the ground, or in the understorey,
3) *Air Foraging*: species that predominantly forage on the wing,
4) *Nectivorous*: species that feed predominantly on nectar in any strata,
5) *Carnivorous*: species that are predominately carnivorous.

7.2.2 Bird Surveys

The methodology for surveying bird diversity in jarrah forest is well established (Craig 1999). All of the twenty sites described in chapter 3.3 were surveyed. Each census consisted of an area search of the 1 ha plot for half an hour (Craig 1999; Craig & Roberts 2001). Only birds utilising the habitat within a plot were included in the data. Swallows, martins and birds of prey flying through the plot were considered to be foraging. No distinction was made between birds that were detected by sight or sound. Nomenclature follows Christidis & Boles

(1994). Appendix 5 has the scientific names of all species recorded in this research.

Sites were surveyed three times in Spring 2000. A seven day period elapsed between surveys (Survey 1: 29/9/2000 - 2/10/2000, Survey 2: 6/10/200 - 10/10/2000, Survey 3: 13/10/2000 - 17/10/2000). All counts commenced at sunrise, with the last count starting no later than 3 hours after sunrise. Time of day, weather and vegetation density in logged and unlogged forest have been shown to have no significant impact on the detection rates of species using this technique (Craig & Roberts 2001). Five surveys were conducted daily, with site order chosen randomly. Problems associated with inter annual variation were removed by surveying birds in all treatments within the same season (noted by Abbott *et al.* 2003). Furthermore, the season of fire was controlled because all burnt forest was disturbed by the same fire (see Chapter 3).

7.2.3 Statistics

The statistical approach in section 5.2.3 was followed.

7.3 Results

7.3.1 General Description

Thirty-nine species of bird were recorded across all treatments (Appendix 5). One species is exotic in southwest Australia (Laughing Kookaburra). The majority of species (n = 14) were 'tree foraging' species or 'ground foraging' (n = 13). Nine of the remaining species were 'nectivorous' (n = 5) or 'air-foraging insectivores' (n = 4). Three species (Brown Goshawk, Whistling Kite, and Southern Boobook) were 'carnivores'. As each of the three species was recorded at only one site, this guild was not assessed in more detail.

Species richness was variable among sites. Fewer than half the species were recorded at each site (range = 11 – 17 spp./site, $\bar{x}$ = 13.7 SE ± 0.4 spp./ site). There was no difference in the species richness of birds between treatments at standard levels of significance ($p < 0.05$). When alpha was adjusted to account for low statistical power (power < 0.8) there was a significant difference among treatments ($p < 0.1$; Table 7.1). Pairwise analysis showed there were more species of bird in forest selectively logged prior to 1950 than both treatments burnt in 2000 ('burnt 2000' and 'gap logged/ burnt 2000') (Fig. 7.1). There were a similar number of species observed in undisturbed forest, forest gap logged in 1990, forest burnt in 2000 and forest gap logged and burnt in 2000 ($p > 0.1$; Fig 7.1).

Table 7.1 One-way ANOVA and power analysis testing whether there are differences in the species richness of birds between treatments representing different disturbance regimes.

	Sum of Squares	Degrees of Freedom	Mean Square	*F* ratio	P Value	POWER
Between groups	0.10	4	0.03	2.74	0.07	0.61
Within groups	0.14	15	0.01			
Total	0.25	19				

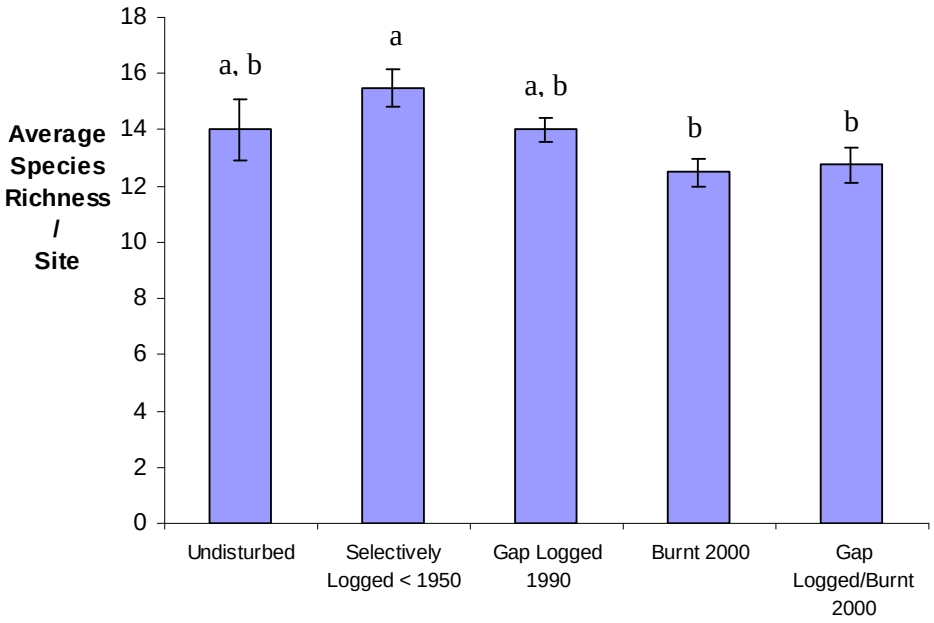


Fig. 7.1 The average (± SE) species richness of birds per site for five treatments representing different disturbance regimes. Significant differences between sites were determined by one-way ANOVA. Different letters indicate significant differences ($p < 0.1$).

Multivariate analysis showed there were no differences in species composition among treatments (Global R = - 0.03; p = 0.61). As seventeen species (48%) were recorded at less than four sites, the composition of species between treatments was variable (Fig. 7.2). There were small differences in composition between sites burnt in 2000 ('burnt 2000' and 'gap logged/burnt 2000') and sites not burnt since 1995 ('undisturbed', 'selectively logged < 1950' and 'gap logged 1990') on the ordination (Fig. 7.2). The composition of species was also more variable in recently burnt sites than unburnt sites (Fig. 7.2). When sites are pooled within treatments, forest gap logged in 1990 and selectively logged prior to 1950 contained more species than unlogged treatments (Fig. 7.3). Undisturbed forest and forest gap logged / burnt in 2000 had equivalent numbers of species (Fig. 7.3). Unlogged forest that was burnt in 2000, had the fewest species (Fig. 7.3).

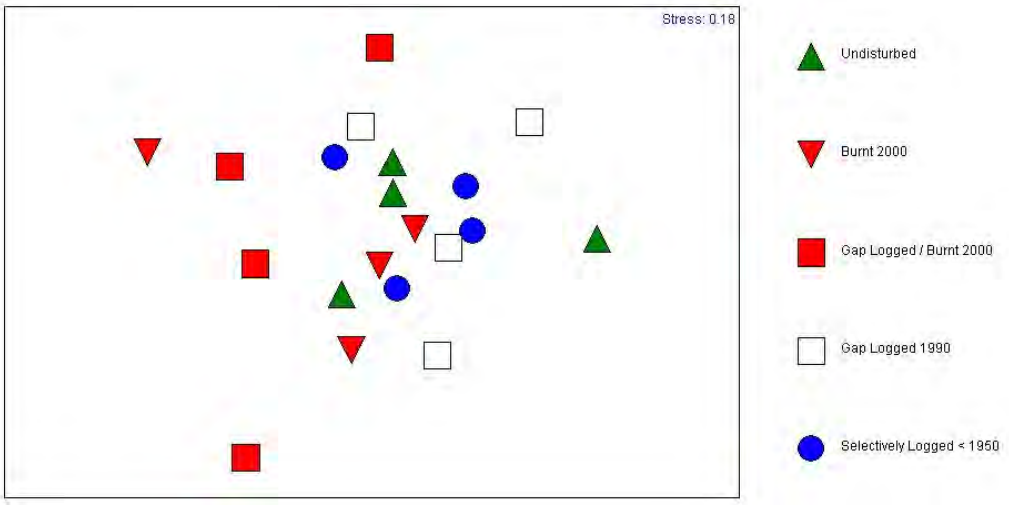


Fig. 7.2 Non-metric multidimensional scaling plot showing the effect of five disturbance regimes on the composition of tree-foraging birds.

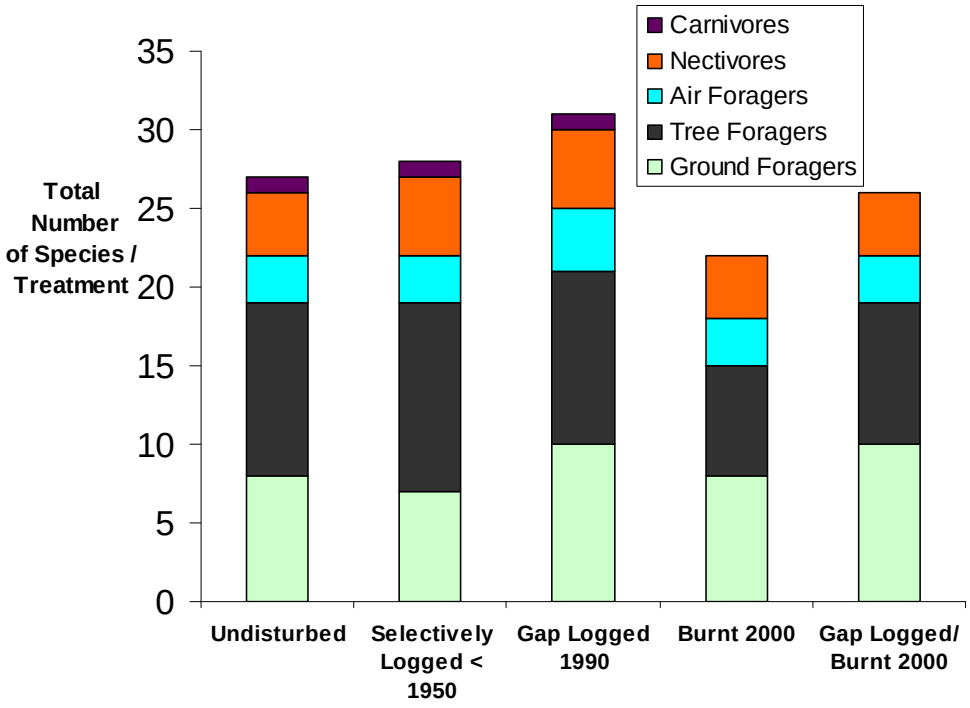


Fig. 7.3 The total number of bird species recorded within each treatment. Each treatment consisted of four sites. Birds are grouped within foraging guilds.

7.3.2 TREE FORAGING SPECIES

Species Richness

There was considerable variation in the number of tree foraging species recorded at each site ($\bar{x}$ = 5.3 SE ± 0.4 spp./site, range = 2-9, spp./site). One species was recorded at all sites (Striated Pardalote) (Table 7.2). When low statistical power was taken into account (i.e., power < 0.8), there were statistical differences in species richness between treatments ($p < 0.1$) (Table 7.3). Pairwise analysis showed that undisturbed forest and forest selectively logged prior to 1950 contained more species than forest gap logged/burnt in (2000) ($p < 0.1$) (Fig. 7.4). The low number of species in forest that was gap logged/ burnt in 2000 was due to two sites (GLB 1 & GLB 3) only containing two species. There were no other differences in the number of species among treatments ($p > 0.1$; Fig. 7.4).

Table 7.2 The tree-foraging birds recorded in five treatments representing different disturbance regimes ('Un' – Undisturbed, 'SL 50' – Selectively Logged < 1950, 'GL 90' – Gap Logged 1990, 'GLB' – Gap Logged/ Burnt 2000, 'B' – Burnt 2000). Numbers represent the number of sites that species were recorded.

FAMILY	SPECIES	Un	SL 50	Gl 90	GLB	B
CACATUIDAE	Red-tailed Black-Cockatoo	0	0	1	0	0
	Carnaby's Black- Cockatoo	0	1	0	1	0
PSITTACIDAE	Australian Ringneck	3	2	3	2	2
	Red-capped Parrot	2	3	2	2	2
CUCULIDAE	Fan-tailed Cuckoo	1	2	2	1	0
	Shining Bronze-Cuckoo	3	3	1	1	4
PARDALOTIDAE	Spotted Pardalote	1	1	1	0	0
	Striated Pardalote	4	4	4	4	4
	Weebill	0	2	1	0	0
MELIPHAGIDAE	White-naped Honeyeater	2	3	1	1	3
NEOSITTIDAE	Varied Sitella	2	1	0	0	1
PACHYCEPHALIDAE	Golden Whistler	3	4	4	1	3
CAMPEPHAGIDAE	Black-faced Cuckoo-Shrike	1	1	1	2	0
ZOSTEROPIDAE	Grey-breasted White-eye	1	0	0	0	0
Total Number of Species / Treatment		**11**	**12**	**11**	**9**	**7**

Table 7.3 One-way ANOVA and power analysis testing whether there are differences in the species richness of tree-foraging birds between treatments representing different disturbance regimes.

	Sum of Squares	Degrees of Freedom	Mean Square	*F* ratio	Significance	POWER
Between groups	1.11	4	0.28	2.83	0.06	0.63
Within groups	1.47	15	0.10			
Total	2.58	19				

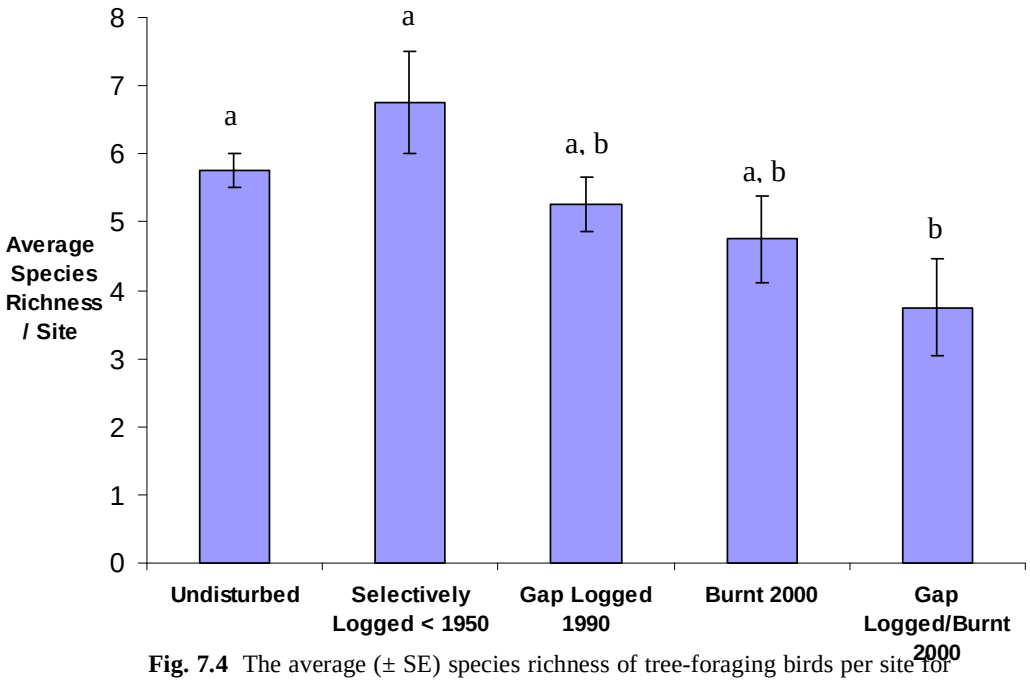


Fig. 7.4 The average (± SE) species richness of tree-foraging birds per site for five treatments representing different disturbance regimes. Different letters indicate significant differences ($p < 0.1$).

Species Composition

No significant difference in tree foraging species composition could be attributed to logging or fire (Global R = -0.01, p = 0.47). There was relatively high variance among sites within treatments which may have contributed to this lack of significant difference in species composition between treatments. Only six species (42%) were recorded across all treatments (Table 7.2). Five other species (Red-tailed Black-Cockatoo, Carnaby's Black-Cockatoo, Weebill, Spotted Pardalotte, and Grey-breasted White-eye) were recorded at less than 4 sites (Table 7.2). These 'uncommon' species were recorded in all treatments except unlogged, burnt forest (undisturbed = 2 spp., gap logged/burnt 2000 = 1 spp., gap logged 1990 = 2 spp., selectively logged prior to 1950 = 1 spp.).

Despite the lack of statistical difference between treatments, six tree-foraging species appeared to be influenced by gap logging. The Varied Sitella was not recorded in forest gap logged in 1990 or in forest gap logged and burnt in 2000, but was recorded in all other treatments (Table 7.2). Shining Bronze-Cuckoo and White-naped Honeyeater were also recorded less frequently in gap logged forest (1990 & 2000) than in other treatments (Table 7.2). The Golden Whistler was only recorded in one site that was gap logged and burnt (2000), but was regularly recorded across the other four treatments (Table 7.2). There was greater similarity in the composition of tree foraging species among sites in undisturbed forest, burnt 2000 and selectively logged prior to 1950 treatments than the two gap logged treatments ('gap logged 1990' and 'gap logged/ treatment 2000') on the ordination (Fig. 7.5). The two gap logged treatments were more variable than the other three treatments (Fig. 7.5).

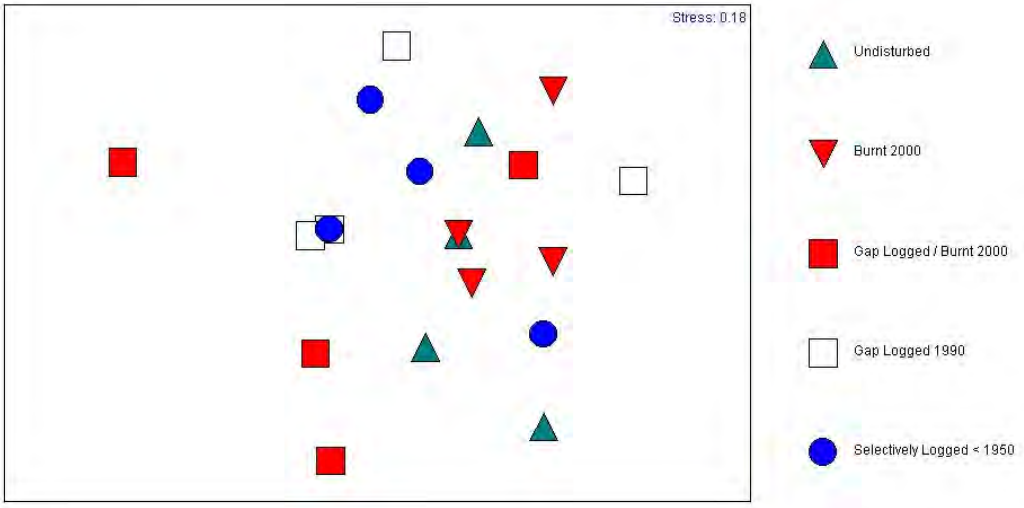


Fig. 7.5 Non-metric multidimensional scaling plot showing the effect of five disturbance regimes on the composition of tree-foraging birds.

7.3.3 GROUND/ SHRUB FORAGING SPECIES

Species Richness

Fewer than half the ground/shrub foraging species were recorded at each site ($\bar{x}$ = 4 SE ± 0.3 spp./site, range = 2- 6, spp./site). There was no statistical difference in species richness between all treatments even when low statistical power was accounted for (Table 7.4). *Post hoc* analysis shows that 32 sites per treatment were required for the ANOVA model to have sufficient power (power > 0.8).

Table 7.4 One-way ANOVA and power analysis testing whether there are differences in the species richness of ground-foraging birds between treatments representing different disturbance regimes.

	Sum of Squares	Degrees of Freedom	Mean Square	F-ratio	P Value	POWER
Between Groups	0.16	4	0.04	0.36	0.83	0.11
Within groups	1.63	15	0.11			
Total	1.79	19				

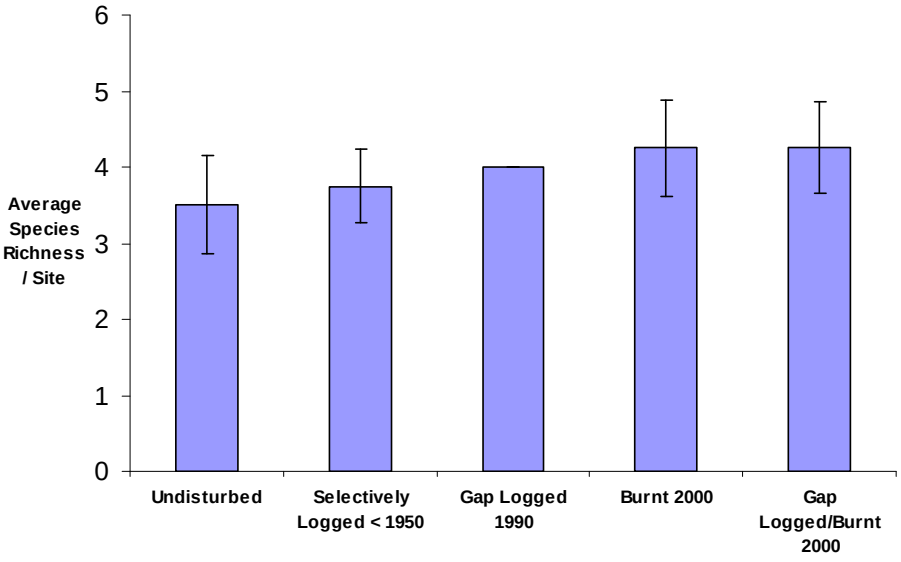


Fig. 7.6 The average (± SE) species richness of ground foraging birds per site for five treatments representing different disturbance regimes.

Community Composition

There was no statistical difference in the composition of ground foraging species between treatments (Global R = - 0.09, p = 0.82). The community of ground

foragers in burnt sites, irrespective of logging, were more similar to each other than with forest that was burnt prior to 1995 (especially the 'undisturbed' treatment) on the ordination (Fig. 7.7). This may have resulted because the Australian Magpie and Australian Raven were only recorded in burnt sites, irrespective of whether sites were logged (Table 7.5). Furthermore, the Scarlet Robin was more frequently recorded in recently burnt forest than in unburnt forest (Table 7.5). None of these three species were recorded in undisturbed forest, and only the Scarlet Robin was recorded in forest selectively logged prior to 1950 and forest gap logged in 1990.

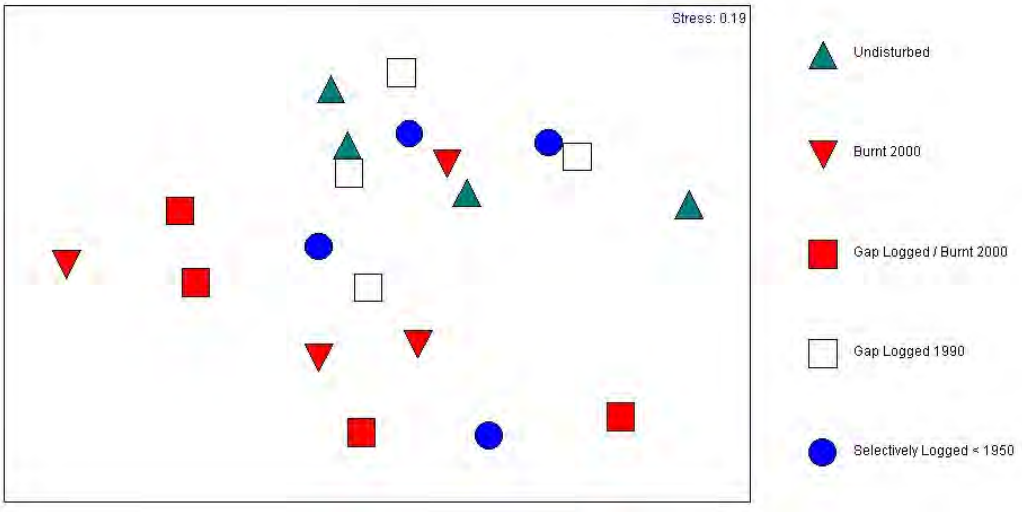


Fig. 7.7 Non-metric multidimensional scaling plot showing the effect of five disturbance regimes on the composition of ground- foraging birds.

Table 7.5 The ground-foraging birds recorded in five treatments representing different disturbance regimes ('Un' – Undisturbed, 'SL 50' – Selectively Logged < 1950, 'GL 90' – Gap Logged 1990, 'GLB' – Gap Logged/ Burnt 2000, 'B' – Burnt 2000). Numbers represent the number of sites that species were recorded.

FAMILY	SPECIES	Un	SL 50	Gl 90	GLB	B
TURNICIDAE	Painted Button-quail	0	0	0	1	0
COLUMBIDAE	Common Bronzewing	1	0	1	1	2
HALCYONIDAE	Laughing Kookaburra	0	2	1	2	2
MALURIDAE	Splendid Fairy-wren	1	1	1	0	0
PARDALOTIDAE	Inland Thornbill	4	4	4	3	3
	Western Thornbill	3	3	4	2	3
	White-browed Scrubwren	1	2	1	0	0
PETROICIDAE	Western Yellow Robin	1	0	1	0	0
	Scarlet Robin	0	1	1	3	3
PACHYCEPHALIDAE	Grey Shrike-thrush	1	0	1	1	0
ARTAMIDAE	Australian Magpie	0	0	0	1	1
	Grey Currawong	2	1	1	2	1
CORVIDAE	Australian Raven	0	0	0	1	2
Total Number of Species / Treatment		**8**	**7**	**10**	**10**	**8**

Several other species had interesting trends. Two species (Inland Thornbill and Western Thornbill) were recorded in all treatments (Table 7.5). In contrast, six species (Painted Button-quail, Splendid Fairy-wren, White-browed Scrubwren, Grey Shrike-thrush, Australian Magpie and Australian Raven) were recorded at less than 4 sites. These 'uncommon' species were distributed evenly across all treatments ('undisturbed' =3 spp., 'burnt 2000' =2 spp., 'gap logged / burnt 2000' = 4 spp., 'gap logged 1990' = 3 spp., 'selectively logged < 1950' = 2 spp.) which may have contributed to the high variability between sites within the same treatment (Fig. 7.7).

7.3.4 NECTIVOROUS SPECIES

Low numbers of nectivores were recorded at all sites ($\bar{x}$ = 2.3 SE ± 1.9 spp./site, range = 1 - 4 spp./site). No significant differences in the number of nectivorous species were found between treatments when low statistical power was taken into account ($p > 0.1$; Table 7.6; Fig. 7.8). *Post hoc* analysis shows that 12 sites per treatment were required for the ANOVA model to have sufficient power (power > 0.8).

Table 7.6 One-way ANOVA and power analysis testing whether there are differences in the species richness of nectivores birds between treatments representing different disturbance regimes.

	Sum of Squares	Degrees of Freedom	Mean Square	F-ratio	P value	POWER
Between Groups	0.66	4	0.17	0.99	0.44	0.24
Within groups	2.48	15	0.17			
Total	3.14	19				

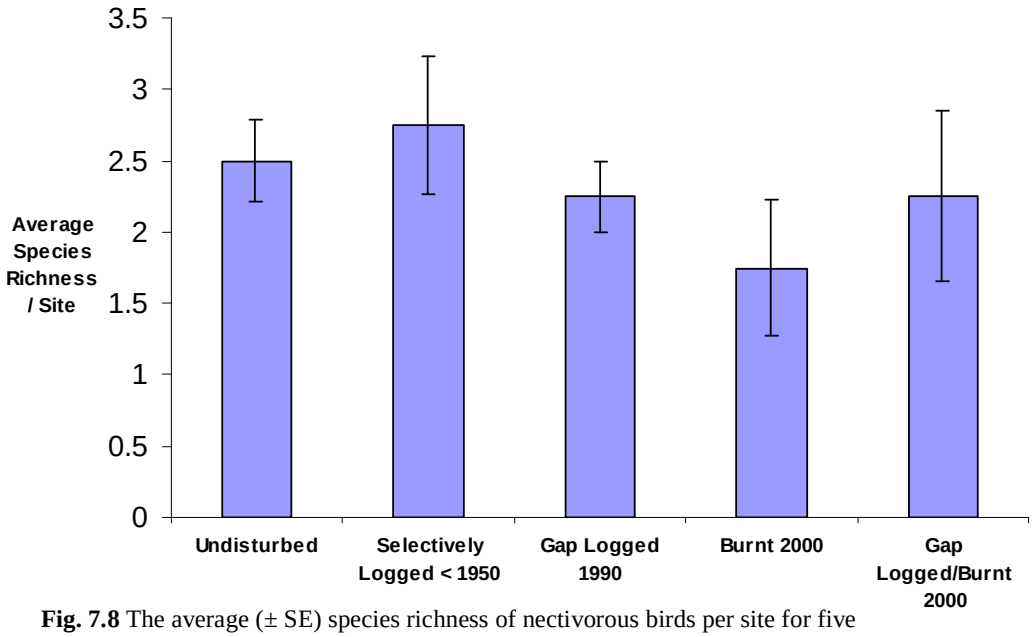


Fig. 7.8 The average (± SE) species richness of nectivorous birds per site for five treatments representing different disturbance regimes.

Community Composition

There were no significant differences in the composition of nectivores between all treatments (Global R = -0.18, p = 0.99). Four of the five species (Brown Honeyeater, Western Spinebill, Little Wattlebird and Red Wattlebird) were found in all the treatments (Table 7.7). Many sites contained the same species (Fig. 7.9). The other species, New Holland Honeyeater, was only recorded in two sites (RL4 & SL1) and therefore contributed minimally to differences.

Table 7.7 The nectivores birds recorded in five treatments representing different disturbance regimes ('Un' – Undisturbed, 'SL 50' – Selectively Logged < 1950, 'GL 90' – Gap Logged 1990, 'GLB' – Gap Logged/ Burnt 2000, 'B' – Burnt 2000). Numbers represent the number of sites that species were recorded.

FAMILY	SPECIES	Un	SL 50	Gl 90	GLB	B
MELIPHAGIDAE	Brown Honeyeater	2	1	1	1	1
	Western Spinebill	3	4	4	3	2
	New Holland Honeyeater	0	1	1	0	0
	Little Wattlebird	2	2	1	2	1
	Red Wattlebird	3	3	2	3	3
Total Number of Species / Treatment		**4**	**5**	**5**	**4**	**4**

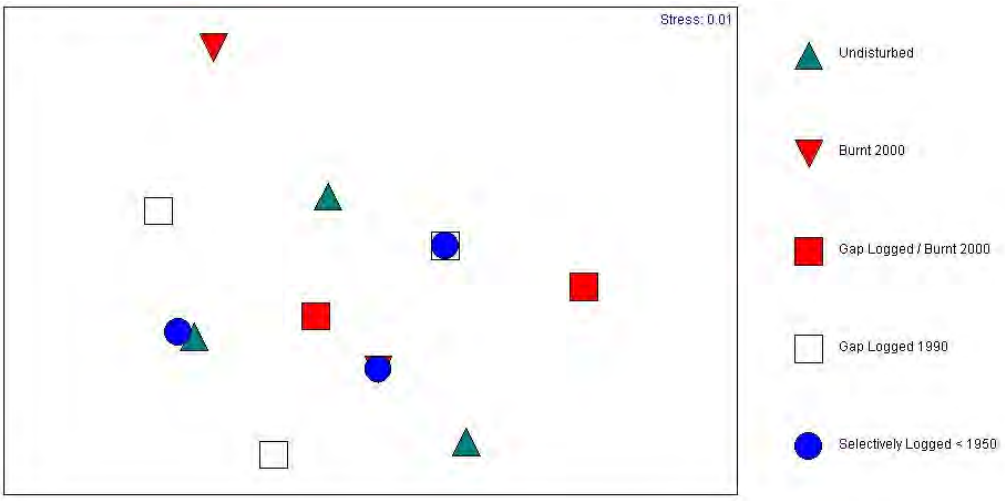


Fig. 7.9 Non-metric multidimensional scaling plot showing the effect of five disturbance regimes on the composition of nectivorous birds.

7.3.5 AIR FORAGING SPECIES

Species Richness

There were low numbers of air foraging species at all sites ($\bar{x}$ = 2.2 SE ± 1.4 spp./site, range = 1-3 spp./site). No difference in species richness was found between treatments ($p < 0.1$; Table 7.8; Fig. 7.10). The one-way ANOVA had low statistical power (power = 0.08). *Post* hoc analysis showed that 50 sites / treatment would have provided sufficient power to test the null hypothesis (power > 0.8).

Table 7.8 One-way ANOVA and power analysis testing whether there are differences in the species richness of air-foraging birds between treatments representing different disturbance regimes.

	Sum of Squares	Degrees of Freedom	Mean Square	F ratio	P Value	POWER
Between Groups	0.09	4	0.02	0.19	0.93	0.08
Within groups	1.80	15	0.12			
Total	1.89	19				

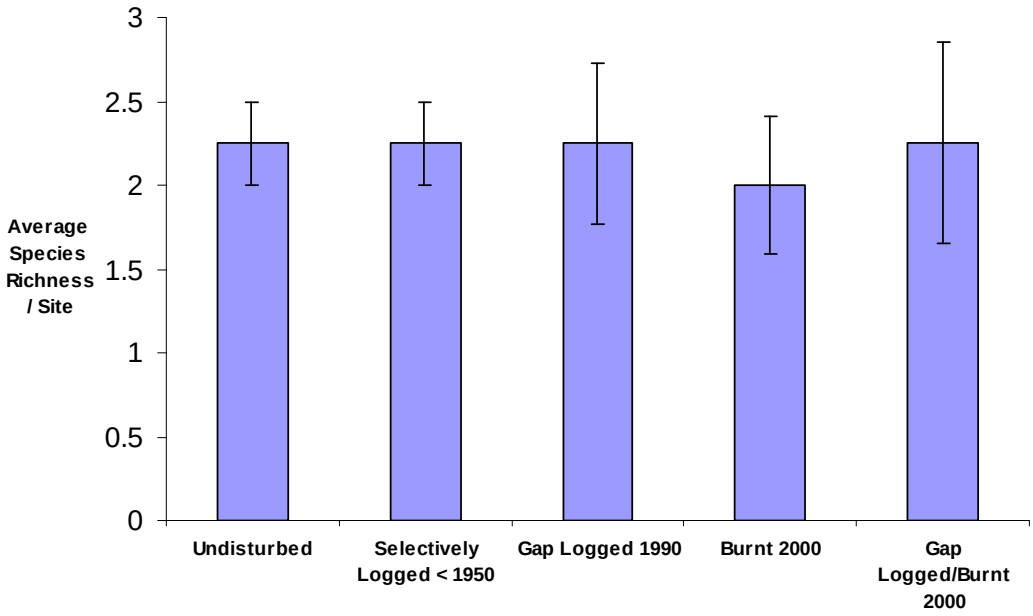


Fig. 7.10 The average (± SE) species richness of air-foraging birds per site for five treatments representing different disturbance regimes.

Community Composition

Despite the low numbers of species found at each site, there were significant differences in the community composition between treatments (Global $R = 0.24$, $p < 0.05$). Pairwise analysis showed that sites that were gap logged and burnt (2000) had a different composition of species than sites in undisturbed forest and in forest selectively logged prior to 1950 (Table 7.9). Two species contributed to the difference in composition between treatments (Table 7.10). Tree Martins were recorded for all gap logged/ burnt (2000) sites but were not recorded in undisturbed forest or selectively logged forest prior to 1950 (Table 7.11). Tree Martins were recorded at one site in burnt (B2) and gap logged 1990 (GL 3) treatments, which resulted in no statistical difference between these treatments and forest gap logged and burnt (2000) (Table 7.9; Table 7.11). Grey Fantails had the opposite trend to Tree Martins, as they were commonly recorded in all treatments except gap logged burnt forest, where it was recorded at one site

(GLB 2) (Table 7.11). The regular occurrence of the other two species meant there was high similarity within sites for all treatments (Fig. 7.11).

Table 7.9 Summary of pairwise analysis from one-way ANOSIM testing the effect of past disturbance regimes on the composition of air-foraging species at Mt Dale. Grey shading denotes significant differences between treatments.

Groups	Global R	Statistic (%)
Undisturbed, Burnt 2000	-0.042	100
Undisturbed, Gap Logged/ Burnt 2000	0.719	2.9
Undisturbed, Gap Logged 1990	-0.188	100
Undisturbed, Selectively Logged <1950	-0.078	100
Burnt 2000, Gap Logged/ Burnt 2000	0.51	8.6
Burnt 2000, Gap Logged 1990	-0.083	100
Burnt 2000, Selectively Logged <1950	0.01	71.4
Gap Logged/ Burnt 2000, Gap Logged 1990	0.547	5.7
Gap Logged/ Burnt 2000, Selectively Logged <1950	0.906	2.9
Gap Logged 1990, Selectively Logged <1950	-0.047	65.7

Table 7.10 The composition of air foraging species that contributed to differences found between treatments identified as being different by ANOSIM analysis. Species listed are those that contributed 5% of the variation as determined by SIMPER analysis (percent contribution). Species are grouped within the treatment they were most commonly encountered.

Treatments	**Species Contributing to Difference**
Undisturbed	Grey Fantail (36.48)
Gap logged / Burnt 2000	Tree Martin (43.68)
Selectively Logged Forest < 1950	Grey Fantail (39.4)
Gap Logged / Burnt 2000	Tree Martin (50.0)

Table 7.11 The ground-foraging birds recorded in five treatments representing different disturbance regimes ('Un' – Undisturbed, 'SL 50' – Selectively Logged < 1950, 'GL 90' – Gap Logged 1990, 'GLB' – Gap Logged/ Burnt 2000, 'B' – Burnt 2000). Numbers represent the number of sites that species were recorded.

FAMILY	**SPECIES**	**Un**	**SL 50**	**Gl 90**	**GLB**	**B**
PARDALOTIDAE	Western Gerygone	3	4	3	4	4
DICRURIDAE	Grey Fantail	4	4	4	1	3
ARTAMIDAE	Dusky Woodswallow	1	1	1	0	0
HIRUNDINIDAE	Tree Martin	0	0	1	4	1
Total Number of Species / Treatment		**3**	**3**	**4**	**3**	**3**

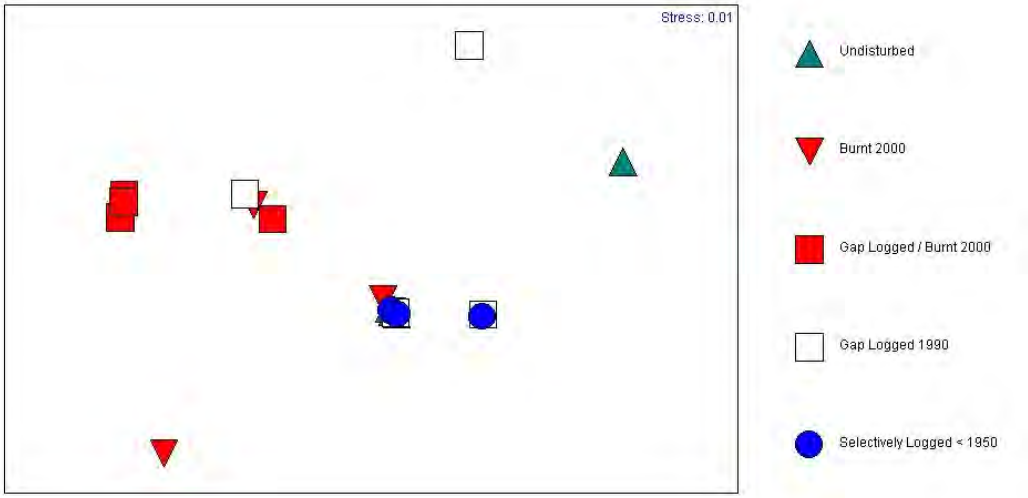


Fig. 7.11 Non-metric multidimensional scaling plot showing the effect of five disturbance regimes on the composition of air-foraging birds.

7.4 Discussion

Changes in bird diversity provide additional support to the assertion that logging practices affect biodiversity for a longer period than fire disturbance. However, in contrast to the findings in the research on understorey plants (chapter 5) and ground spiders (chapter 6), more species were recorded in forest selectively logged prior to 1950 compared with that in undisturbed forest or forest gap logged in 1990. Differences in the diversity of different foraging guilds show that changes to tree, air and ground foraging resources occur for at least 10 years after gap logging. These changes are well beyond the temporal changes to bird diversity caused by a single fire event, which is expected to cause changes for a maximum of seven years given the rotation length between prescription fires.

Unlogged forest contained fewer species of bird than logged forest (gap logged /burnt 2000, gap logged in 1990, and selectively logged prior to 1950). This is not surprising, as the vast majority of species that forage in jarrah forest are considered 'adaptable' and 'opportunistic' (Abbott & Van Huerck 1985b; Wykes 1985; Norwood *et al.* 1995; Abbott 1999). Most species in jarrah forest are widely distributed throughout southern Australia in non-forest habitats (Craig 1999). In Western Australian forests, more than 90 species are believed to have benefited from the provision of new habitats resulting from logging and land clearance since European settlement, compared with 6 species that are considered to have been deleteriously affected (Lewin's Rail *Rallus pectoralis*, Noisy Scrub-Bird *Atrichornis clamosus*, Malleefowl *Leipoa ocellata,* Red-tailed Black Cockatoo *Calyptorhynchus banksii*, Western Corella *Cacatua pastinator*, and Rufous Treecreeper) (Abbott 1999). Of the species deleteriously affected

since European arrival, only Red-tailed Black Cockatoos were recorded at Mt Dale. This species was recorded in forest that had been gap logged in 1990.

Norwood *et al.* (1995) suggested that it is possible that previous disturbance history might have already caused reductions in the distribution and abundance of species dependent on mature forest. Thus, the bird communities that now occur in jarrah forest are composed of 'generalists' (those species that may benefit from logging) that have replaced 'specialists' that may have been adversely affected by logging. At Mt Dale, all species of bird that were recorded in unlogged forest were also recorded in production forests (e.g., forest disturbed by contemporary logging or past selective logging), lending support to this possibility. In other words, no species were 'dependent' on unlogged forest at Mt Dale because they have already become locally extinct in the area. Although theoretically possible, this hypothesis is now difficult to test because there are no pre-logging bird data, and as almost all productive jarrah forest has either been logged, or is within close proximity of forest that has being logged (as noted by Norwood *et al.* 1995).

7.4.1 The Impact of Fire in Unlogged and Logged Forest

The composition of species was similar in the two treatments that experienced fire in 2000 (unlogged/burnt forest and gap logged/burnt forest). Three species appear to be positively affected by recent fire. The Australian Magpie and Australian Raven are transient, non-dependent residents of forests of the southwest, and have been recorded in other studies as increasing in numbers after recent fire (Kimber 1974b; Abbott 1999). These species forage widely on disturbed habitat in rural and urban environments, and therefore it is not

surprising that they utilise recently burnt jarrah forest. A third species (Scarlet Robin) was more common in burnt than undisturbed forest at Mt Dale. In other regions of south-western Australia, Scarlet Robins have been shown to become abundant in forest and woodlands where fire has removed the vegetation (Burbidge 2003).

Four other species showed reduced occurrence in both recently burnt treatments. Three of these species (Splendid Fairy-wren, Western Yellow Robin and White-browed Scrubwren) require complex leaf litters and understorey vegetation (Abbott 1999), which is habitat that was destroyed by the fire at Mt Dale (Chapter 4). Although the fourth species (Grey Shrike-thrush) is ubiquitous in most ecosystems, this species has been shown to be negatively affected by fire in other ecosystems (e.g., Chapman & Newby 1994). After moderate intensity fire, other studies in jarrah forest have shown that changes in species assemblages, abundances, and diets, may be affected for several years (Kimber 1974b; Christensen ***et al.*** 1985; Wooller & Calver 1988). Within this period, open ground foragers are replaced by species that forage within developing understorey vegetation and leaf litter.

There were small differences in the composition of species between the two burnt treatments. The Tree Martin was more frequently recorded in gap logged/ burnt forest than in unlogged/burnt forest. This is an interesting finding, because although Tree Martins prefer to forage in open areas (Abbott 1999), they have been found to be negatively affected by logging in other temperate forest (e.g., Loyn 1979). This is because Tree Martins nest in hollow trees, the density of which can be substantially reduced following logging. In contrast to Tree

Martins, Grey Fantails were recorded more frequently in unlogged/burnt forest than in gap logged/burnt forest. This supports some previous findings that show that Grey Fantails are affected by contemporary logging practices (Norwood *et al.* 1995). As Grey Fantails utilise tree branches to perch and forage from, the loss of trees that result from gap logging is a possible cause in the reduction of these species in recently gap logged sites (Craig 1999).

Reductions in four tree foraging species (Varied Sitella, Shining Bronze-cuckoo, White-naped Honeyeater, and Golden Whistler) were also recorded in forest gap logged/burnt forest but not in unlogged/burnt forest. White-naped Honeyeaters and Golden Whistlers have previously been identified as species that are negatively affected by logging disturbance (Craig 1999). As was noted for the Grey Fantail, each of these species forages in trees (the tree-foraging guild), the loss of trees that results from gap logging is the most likely cause in the reduction of these species in recently gap logged sites. This assertion is supported by studies in other temperate forests that show that intensive logging affects species that forage in trees (e.g., Loyn 1979; Thompson *et al.* 1999; Williams *et al.* 2001).

7.4.2 Recovery of Bird Diversity 10 years after Gap Logging

Unlike understorey plants and ground spiders, a similar number of bird species were recorded in forest gap logged in 1990 and undisturbed forest. One possible reason for this similarity is that the regeneration of jarrah and marri (as well as other ground vegetation) following disturbance provides foraging habitat for almost all ground and tree foraging insectivores (as shown by Wykes 1985; Armstrong & Nichols 2000). Although mature trees are removed by logging

(Chapter 4; Calver & Dell 1998a), both overstorey and total cover exceed 80% of the value for unlogged forest within five years (Stoneman *et al.* 1989). At Mt Dale, there was no difference in horizontal canopy cover and understorey cover between sites that were undisturbed and sites that were gap logged in 1990 (Chapter 4). Given that regenerating trees can contain more arthropods and lurp (Wykes 1985; Armstrong & Nichols 2000), and are also capable of flowering (pers. obs.), forest recovering from logging disturbance may provide similar foraging resources as unlogged forest.

Some species appeared to be negatively affected by gap logging. Three tree foraging species (Varied Sitella, Shining Bronze-cuckoo and White-naped Honeyeater) were not recorded in forest that was gap logged forest in 1990 as regularly as they were in unlogged forest or forest selectively logged before 1950. All three species forage predominantly in mature trees (Abbott & Van Huerck 1985a; pers. obs) and are therefore likely to be affected for an extended period of time in gap logged forest. The height of the trees is predicted to match that of unlogged forest after *c.* 40 years (Abbott *et al.* 2003), whereas the diameter distribution of trees is expected to take centuries (Chapter 4). In other *Eucalypt* dominated forest, disturbance that is similar to that in gap logging (e.g., clearfelling) can affect the composition of species for several decades (e.g., Greenberg *et al.* 1995; Loyn 1998; Thompson *et al.* 1999; Williams *et al.* 2001; Kavanagh & Stanton 2003).

7.4.3 The Diversity of Birds in Forest Selectively Logged Prior to 1950

There were more species in forest selectively logged prior to 1950 than in undisturbed forest. Although the diversity (species richness and composition) of all guilds was similar between these two treatments, there appeared to be more tree foraging species in forest selectively logged prior to 1950 than in undisturbed forest. However, the species identified as being susceptible to the immediate effects of logging in this and earlier studies (see descriptions of Abbott & Van Huerck 1985a; Norwood *et al.* 1995; Craig 1999; Abbott *et al.* 2003) were able to forage in the mature trees found in these previously selectively logged forest at Mt Dale. My research therefore shows that although some structural attributes have not recovered from previous selective logging practices (Chapter 4), these disturbed stands appear to provide similar foraging resources as unlogged forest. If these trends were consistent throughout the jarrah forest, a landscape consisting of selectively logged forest and unlogged forest would contain a similar number of species as unlogged forest, and is likely to contain more species than a landscape mosaic comprising unlogged forest and forest disturbed by gap logging in the last 10 years.

7.4.4 Future Research

There are several outcomes of this research that need to be considered in future studies. The first relates to limitations with regards to measuring the species richness of bird communities. Differences between treatments in the species richness of tree foragers (Fig. 7.2) and nectivores (Fig. 7.4) might exist but the ANOVA did not have the required statistical power to adequately test the null hypothesis. As was also articulated in Section 5.4.4, measuring species richness

requires at least 8 sites per treatment to gain the statistical power required to test whether there are differences in species richness between treatments.

The second outcome refers to the methodology used in this research to determining which species are occurring in each of the different treatments. There are three reasons why future studies that aim to assess how logging affects bird communities should not use the same methods as used in this study. Firstly, most of the species in the jarrah forest are opportunistic foragers, and will incorporate recently disturbed areas into their foraging habitat even if logging deleteriously impacts upon other resources (e.g., breeding habitat). Secondly, all species in the jarrah forest have territories that are larger than the area disturbed by gap logging (which is a maximum of 10 hectares) because they have evolved within an ecosystem that has temporary and spatially patchy distributed resources. As such, species may be only moving through a recently logged area to get to another area, but are registered as occurring (and therefore utilising) this habitat. Thirdly, this methodology does not adequately assess the bird communities that occur in different stands of forest. For example, species that were not recorded in undisturbed forest in this study included Red-tailed Black Cockatoo, Weebill, Scarlet Robin, Grey Shrike-Thrush, Grey Currawong, Black-faced Cuckoo-shrike and Fan-tailed Cuckoo. These species have been recorded occurring in unlogged forests in other studies (see Wykes 1985; Norwood *et al.* 1995; Craig 1999; Abbott *et al.* 2003). I believe these species were not recorded because they exist at low density in the jarrah forest, and were not in any of the unlogged sites when sampling was being undertaken.

As birds occur at low density throughout the jarrah forest (Abbott 1999), rigorous surveys over much larger areas than one hectare are required to accurately describe the bird community. For example, in wet sclerophyll forest in Victoria, MacNally (1997) showed that the surveys in the range of 50 ha were the most appropriate to deliver the power needed to accurately assess differences in community diversity. As this type of survey is not applicable to the scale associated with logging disturbance (maximum of 10 ha for gap logging), other techniques need to be used to assess how birds are affected by contemporary disturbances.

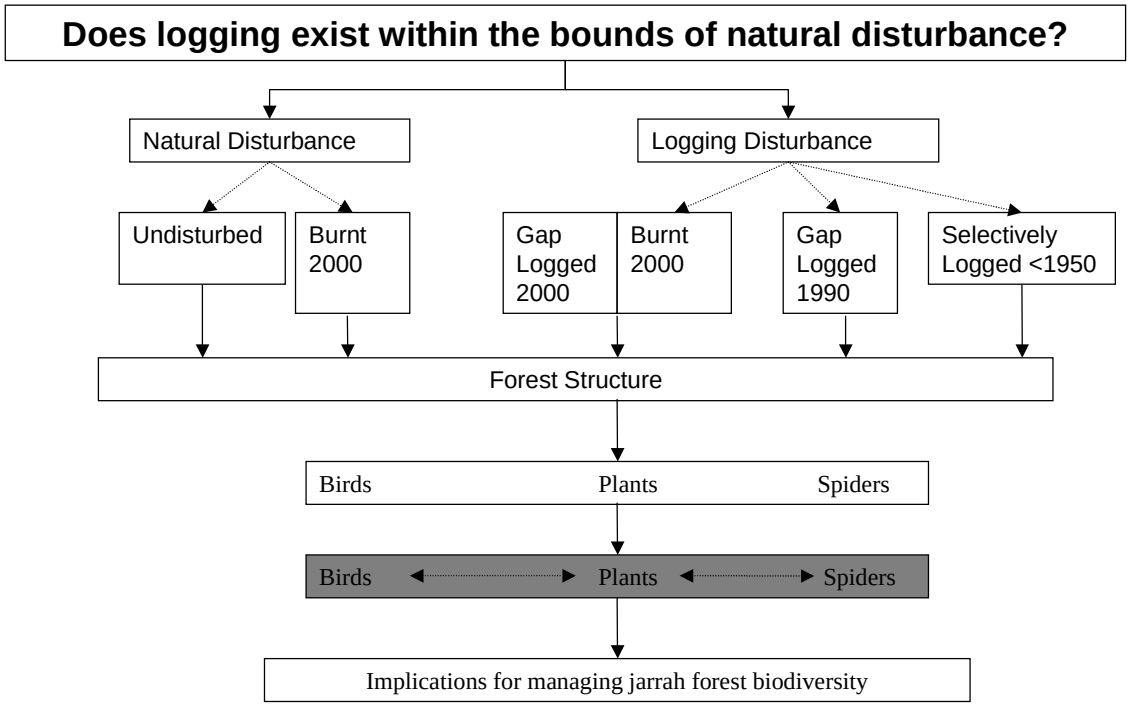
An alternative to measuring community diversity is to measure the effects of logging and fire on individual species (e.g., Bourque & Villard 2001; Robinson & Robinson 2001; Lee *et al.* 2002). The influence of stand (e.g., habitat trees), patch (e.g., buffer strips) and landscape (e.g., the proportion of land and severity of logging disturbance) parameters could be investigated in relation to the populations of specific species. Studies should focus on the breeding biology and population of species across these spatial scales, as this will ultimately determine their long-term viability. Furthermore, ongoing monitoring on specific species will allow for a greater understanding of the inter and intra annual movement patterns that exist in bird populations in jarrah forest (Abbott 1999; Abbott *et al.* 2003). In addition to the species identified by Abbott *et al.* 2003 as being candidates that are important to study (i.e., Western Gerygone and Inland Thornbill), I recommend that future research be conducted on the three tree foraging species that could be negatively effected by gap logging (i.e, Varied Sitella, Shining Bronze-cuckoo and White-naped Honeyeater)

7.4.5 Conclusion

Bird communities respond differently to contemporary logging compared with ground spiders and understorey plants. The total number of bird species increased 10 years after gap logging disturbance, whereas understorey plant diversity and ground spider diversity decreased. Some species were more regularly recorded in forest disturbed by gap logging in 1990 than other treatments. However, gap logging appears to negatively affect other bird species (predominantly tree foraging species) for at least 10 years. Forest selectively logged prior to 1950 had had very similar composition of species as undisturbed forest. Differences between the scale at which birds utilise jarrah forest (greater than 10 ha), and the scale at which gap logging occurs (less than 10 ha), means that future investigations should assess the individual requirements of species, rather than the diversity of species found within a patch.

CHAPTER 8

Surrogacy Among Taxa



Chapter 8

8.1 Introduction

Understanding relationships in species diversity among taxa has important implications for conserving forest biodiversity (Noss 1990; Humphries *et al.* 1995; Ehrlich 1996; Howard *et al.* 1998; Pharo *et al.* 1999). As the resources, techniques, and expertise needed to survey biodiversity are often lacking (Kati *et al.* 2004), broad-scale bioassessment and conservation evaluation is usually based on measuring the diversity of representative taxa (Williams & Gaston 1994). These taxa are termed 'biodiversity indicators' because their diversity is used as a surrogate for all the biodiversity found within an ecosystem (Caro & O'Doherty 1999). The idea is that biodiversity indicators have cross-taxon congruent patterns of species richness (α diversity) (Kati *et al.* 2004) and community similarity (β diversity) (Su *et al.* 2004) with other taxa. In forests, well-known taxonomic groups including trees (Margules & Stein 1989; Foody & Cutler 2003), mammals (Mittermeier 1988; Medeillin *et al.* 2000), birds (Canterbury *et al.* 2000), butterflies (Kremen 1992) and plants (Pharo *et al.* 1999) have been used as biodiversity indicators. If there is congruence among the diversity of different taxa, biodiversity indicators can be used to facilitate the conservation of all taxa, and in doing so, conserve the genetic diversity they contain and the ecological processes they maintain.

Many strategies designed to conserve biodiversity in Australian forests assume that measurements of α and β diversity of a few taxa adequately represent overall trends in biodiversity. These include the current principles used to design a Comprehensive, Adequate and Representative (CAR) reserve system (JANIS 1997; Section 2.5.1) and principles that aim to achieve ecologically sustainable

forest management (ESFM). In jarrah forest, conservation reserves and silvicultural practices are based on relatively few studies that have focused on birds (e.g., Abbott & van Huerck 1985b; Norwood *et al.* 1995; Craig 1999), mammals (e.g., Morris *et al.* 1996), understorey plants (e.g., Burrows *et al.* 2002b), and some arthropod groups (e.g., Strehlow *et al.* 2002; Abbott *et al.* 2002). Almost nothing is known of the life history, distribution or diversity of most species (e.g., as shown by Brennan (2002) for spiders and Judd (2004) for Isopods), or how these organisms are affected by contemporary anthropogenic disturbances (e.g., logging, prescribed fire or mining). As such, relatively well-studied taxa have become surrogates for biodiversity by default. Although the CAR reserve system and ESFM aims to ensure the ongoing viability of all biodiversity, it assumes available information for a few taxa will provide the basis to do this. Three relatively well-studied taxa (understorey plants, ground spiders and birds) when analysed separately have responded differently to logging and fire disturbance at Mt Dale (Chapters 5, 6 & 7). By making the assumption that all species are equal among these taxa, a comparison of the effects of fire and logging on the 'total biodiversity' measured in this research was assessed by addressing three questions:

1) Are there differences in the richness or assemblage of all species in forest recently disturbed by fire compared with forest recently disturbed by gap logging and fire?

2) Does the richness or assemblage of all species recover 10 years after gap logging?

3) Does the richness or assemblage of all species recover 50 years after selective logging?

Despite their widespread use, the relationships between indicators and biodiversity are not well established (Lindenmayer *et al.* 2000; Catteral *et al.* 2004). Few researchers have compared patterns of cross-taxon congruency in species richness and species composition. Those that have assessed cross-taxon congruency usually find weak relationships among taxa (Kati *et al.* 2004). Evidence suggests that the correlation of species richness and coincidence of diversity 'hotspots' between pairs of taxa is variable (Prendergast *et al.* 1993). In temperate forests in New South Wales, Oliver *et al.* (1998) showed that α and β diversity were not generally correlated among ground active invertebrates (ants and beetles), vascular plants and vertebrates (birds, small mammals, frogs, and reptiles). Likewise, Pharo *et al.* (1999) found relatively weak correlations in α and β diversity between vascular plants, lichens and bryophytes across sites in coastal lowland forest in New South Wales. It is imperative that among-taxon congruence in biodiversity patterns is understood, otherwise assessments based on a few well-known groups may lead to erroneous conclusions about the biodiversity patterns of other taxonomic groups, resulting in entirely inappropriate conservation measures (Paavola *et al.* 2003).

The adequacy of one taxonomic group representing the diversity of other taxa has never been assessed in jarrah forest. Given that surrogacy is an important assumption in conservation strategies, and that there is increasing evidence that in other temperate forests surrogacy is not evident, understanding cross-taxon congruency among taxa is required. In the research presented here, cross-taxon congruency was assessed using assemblage fidelity (defined as the degree to which assemblages from different phylogenetic groups co-occur in space and time) (*sensu* Oliver *et al.* 1998). All taxa were measured at similar times,

allowing for a comparison of the assemblage fidelity for the three different taxa across local spatial scales. This chapter will therefore address two additional questions:

1) Are there relationships in α diversity among understorey plant, bird and ground spider taxa among different sites at Mt Dale?
2) Are there relationships in β diversity among understorey plant, bird and ground spider taxa among different sites at Mt Dale?

8.2 Methods

8.2.1 The Effects of Logging and Fire on 'Total Biodiversity'

The methods for sampling the diversity of understorey plants (section 5.2.2), ground spiders (section 6.2.2) and birds (section 7.2.2) are described in their respective chapters. 'Total' biodiversity for each site was estimated by combining the three datasets of understorey vegetation, ground spiders and birds. 'Total' species richness was calculated by adding the species richness of each of three taxa found at each site. Similarly, compositional changes were assessed by combining the three datasets that detailed the presence/absence of each species at each site. A description of the statistics used to assess differences in species richness (ANOVA) and species composition (ANOSIM and nMDS) among the five treatments is given in section 5.2.3.

8.2.2 Surrogacy Among Taxa

Species Richness

The relationship between species richness between two of the three taxa (i.e., understorey plants versus spiders; understorey plants versus birds; birds versus understorey plants) were analysed by correlation using the SPSS package. Spearman rho statistics were used because correlations could be non-linear. The only statistical assumption within this test is to ensure that the data come from a bivariate normal distribution (i.e., that both data sets come from a normal distribution). Normality was tested using the SPSS statistical package. Since outlying data points can have significant impact on simple linear correlations, any outliers after the initial test were removed, and a *post hoc* test undertaken. Appropriate caution was applied on interpretations after *post hoc* removal of any data.

Community Composition

Species turnover can be defined in the following overlapping senses: (i) turnover along gradients, (ii) turnover through space, and (iii) a measure of the difference between samples (Ricklefs & Schluter 1993). In this research, species turnover was used in the second sense (turnover through space) because changes in composition were measured between 20 sites simultaneously at the one time. Comparisons of compositional change among the three taxa were achieved by investigating relationships in similarity indices. Using the PRIMER statistical package, nMDS ordinations were conducted on each of the three taxa using Bray-Curtis dissimilarity matrices. Bray-Curtis matrices were used because this procedure has many properties amenable to ecological data, including independence from scale of measurement and from joint absences (Su *et al.* 2004). Relationships between each similarity matrix were analysed using the Spearman rank co-efficient (as recommended by Daniel 1991). The correlation co-efficient (ρ) lies in the range (-1 to 1), with the extreme of p=-1 and +1 corresponding to the cases where two sets of ranks are in complete opposition or in complete agreement (Clarke & Gorley 2001). The RELATE statistic was used to assess whether similarities in community composition between sites was statistically similar.

8.3 Results

8.3.1 The Effects of Logging and Fire on 'Total Biodiversity'

Species Richness

There were differences in the average number of species per site among the five treatments at standard levels of significance ($p < 0.05$; Table 8.1). Pairwise analysis showed that more species were recorded in undisturbed forest than forest that was gap logged in 1990, forest gap logged and burnt in 2000 and forest burnt in 2000 ($p < 0.05$; Fig. 8.2). Pairwise analysis also showed there were less species in the two treatments burnt in 2000 ('gap logged/burnt 2000' and 'burnt 2000') than treatments representing forest gap logged in 1990 and selectively logged prior to 1950 ($p < 0.05$; Fig. 8.2). In contrast, there were no differences in the average number of species between undisturbed forest and forest selectively logged prior to 1950 ($p > 0.05$; Fig. 8.2). There were also no differences in the number of species in forest that was gap logged and burnt in 2000 and forest burnt in 2000 ($p > 0.05$; Fig. 8.2).

Table 8.1 One-way ANOVA and power analysis testing whether there are differences in the species richness for all taxa between treatments representing different disturbance regimes.

	Sum of Squares	Degrees of Freedom	Mean Square	F-ratio	P Value	POWER
Between Groups	.183	4	.046	9.070	.001	0.99
Within Groups	.076	15	.005			
Total	.258	19				

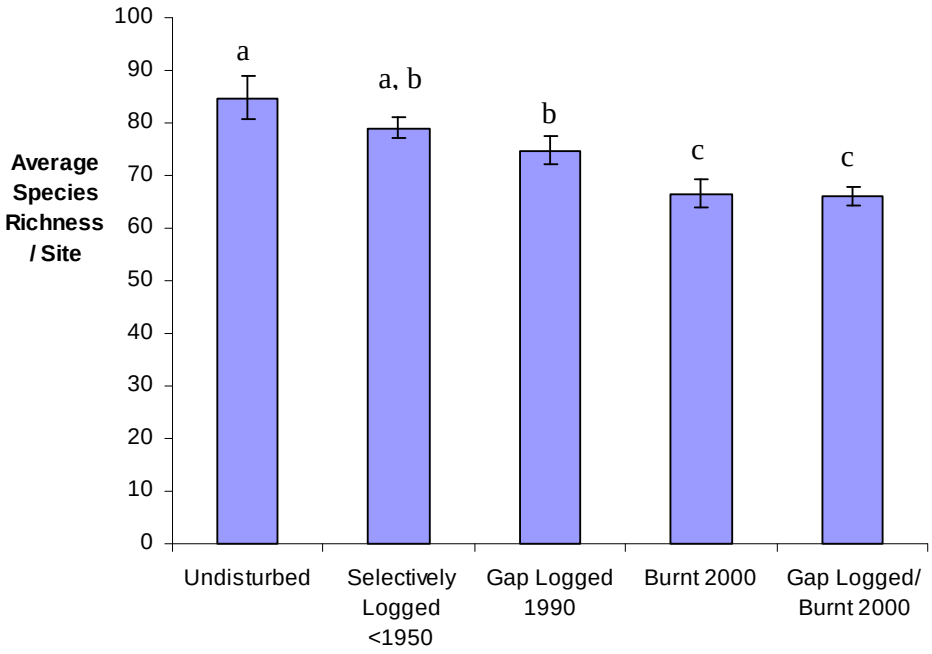


Fig. 8.2 The average (± SE) species richness of all species (understorey plants, ground spiders and birds) for five treatments representing different disturbance regimes. Significant differences between sites were determined by oneway ANOVA. Different letters indicate significant differences ($p < 0.05$).

Species Composition

Analysis of Similarity showed that there were differences in the composition of understorey species among treatments (Global $R = 0.41$, $p < 0.01$). Pairwise analysis showed that there were significant differences between all treatments with two exceptions ($p < 0.05$; Table 8.2). The first was the composition of species within sites that were burnt in 2000 was similar to the composition of species in sites that were gap logged and burnt in 2000 ($p = 0.086$; Table 8.2). The second exception was the composition of species within sites gap logged in 1990 was similar to forest that was selectively logged prior to 1950. In fact, there was much greater similarity between these two treatments than between any other pair of treatments ($p = 0.743$; Table 8.2).

Table 8.2 Summary of ANOSIM pairwise analysis testing whether the composition of all species (understorey plants, ground spiders and birds) were different between five treatments representing different disturbance regimes. Highlighted rows denote significant differences between treatments ($p < 0.05$).

Comparison between Treatments	*R* statistic	P Value
Undisturbed, Burnt 2000	0.43	0.029
Undisturbed, Gap Logged/ Burnt 2000	0.80	0.029
Undisturbed, Gap Logged 1990	0.52	0.029
Undisturbed, Selectively Logged <1950	0.53	0.029
Burnt 2000, Gap Logged/ Burnt 2000	0.34	0.086
Burnt 2000, Gap Logged 1990	0.48	0.029
Burnt 2000, Selectively Logged <1950	0.60	0.029
Gap Logged/ Burnt 2000, Gap Logged 1990	0.28	0.086
Gap Logged/ Burnt 2000, Selectively Logged <1950	0.45	0.029
Gap Logged 1990, Selectively Logged <1950	-0.14	0.743

There were marked differences in the composition of species found in different treatments on the ordination (Fig. 8.2). The composition of species in the four undisturbed sites was different to all other treatments (Fig. 8.2). Similarly, the composition of species in sites that were burnt in 2000 was different to the other four treatments (Fig. 8.2). In contrast, the composition of species in the three treatments disturbed by logging ('selectively logged < 1950', 'gap logged 1990' and 'gap logged/ burnt 2000') appeared to have much greater similarity in species composition with each other than they do with sites that were undisturbed or burnt in 2000 (Fig. 8.2). Furthermore, sites within each of these three treatments occurred in a much smaller area of the ordination than sites in either undisturbed forest or forest burnt in 2000, indicating there was greater similarity in species composition between sites that have been disturbed by logging (Fig. 8.2).

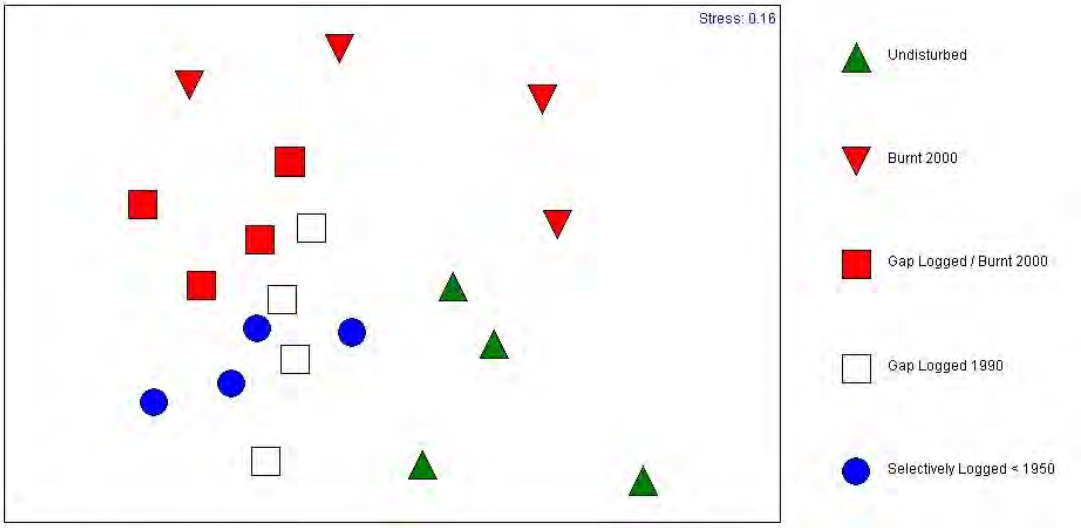


Fig. 8.2 Non-metric multidimensional scaling plot showing the effect of five disturbance regimes on the composition of all species (understorey plants, ground spiders and birds).

8.3.2 Cross-taxon Congruence

Species Richness

Patterns of cross-taxon species richness were variable among taxa and study sites (Table 8.3). There were five times more species of plant than birds or spiders at each site. The strongest relationship was between plants and birds and was not significant (r^2 = 0.26; p > 0.05) (Fig. 8.3). Similarly, the relationship between plant and spider species richness (r^2 = 0.16; p > 0.05), and bird and spider species richness (r^2 = 0.07; p > 0.05), was not significant (Fig. 8.3).

Table 8.3 Summary statistics for bird, understorey vegetation and ground spider species richness across 20 sites at Mt Dale.

Variable	Mean	SD	Minimum	Maximum
Birds	13.7	1.6	11	17
Understorey Plants	58.7	8.0	49	84
Ground Spiders	8.7	3.4	3	14

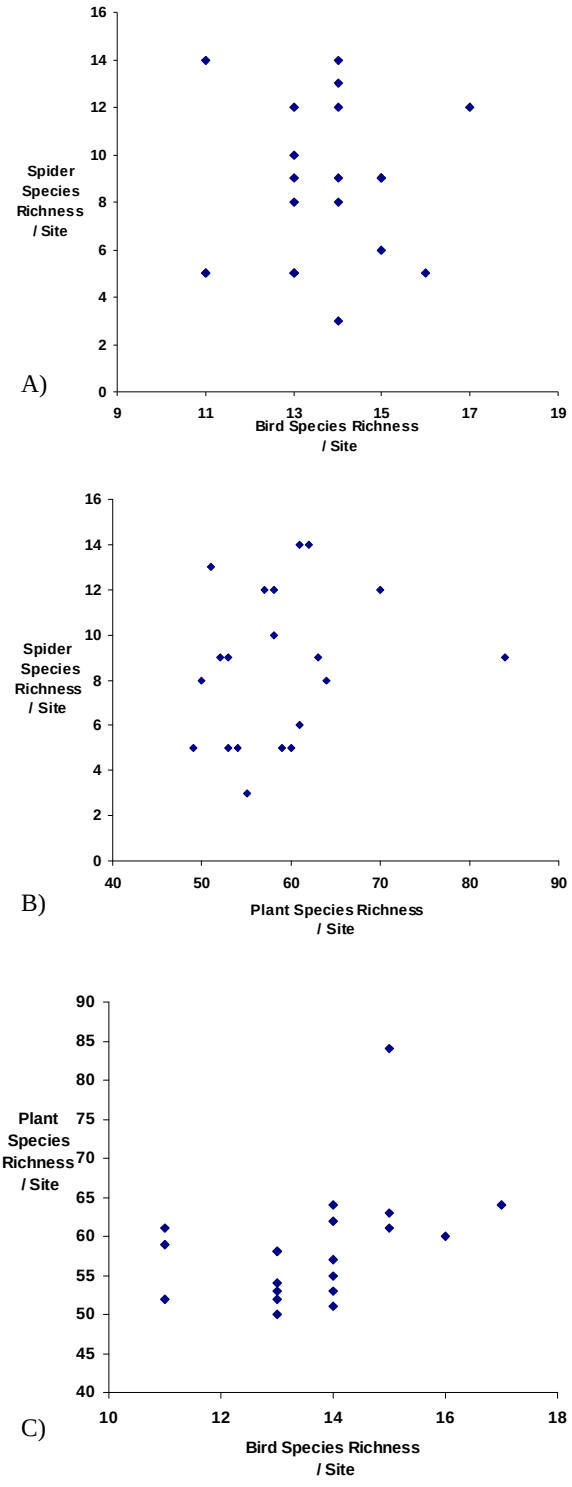


Fig. 8.3 The relationship between the number of species of (a) ground spiders and birds, (b) ground spiders and understorey plants, and, (c) understorey plant species and birds, between 20 sites at Mt Dale.

The lack of relationship may have resulted because one site ('U2') had a much larger number of plant species than other sites. However, *post hoc* removal of this site did not yield significant relationships between the number of understorey plants and ground spiders ($r^2 = 0.09$, $p > 0.05$) or understorey plants and birds ($r^2 = 0.16$, $p > 0.05$).

Species Composition

There were no significant relationships in β diversity among the three taxa ($p < 0.05$). Changes in the composition of birds were not related to changes in the composition of ground spiders ($\rho = 0.19$) than understorey plants ($\rho = 0.13$). Similarly, there was no relationship between ground spiders and understorey plants ($\rho = 0.11$).

8.4 Discussion

There were differences in the total diversity of species among the five treatments. With respect to species richness, the two treatments burnt in 2000 ('burnt 2000' and 'gap logged/burnt 2000') had fewer species than the three treatments that were last burnt before 1995 ('undisturbed forest', 'selectively logged < 1950', and 'gap logged 1990'). Moreover, there were fewer species in forest that was gap logged in 1990 than in undisturbed forest. There were similar numbers of species between the two treatments burnt in 2000, and between sites in undisturbed forest and in forest selectively logged prior to 1950. Importantly, changes in species assemblages among treatments did not reflect changes in species richness. The assemblage of species in undisturbed forest was different to all other treatments, including forest selectively logged prior to 1950. There was also greater similarity in the composition of species between forest selectively logged prior to 1950 and gap logged in 1990 than between forest selectively logged prior to 1950 and undisturbed forest. These results confirm that logging disturbance does not mimic fire disturbance because changes to total biodiversity occur for a much greater period of time (i.e., greater than 50 years) after logging disturbance than after fire disturbance (i.e., 5 – 7 years).

This research also showed there were low levels of congruence in α and β diversity among understorey plants, ground spiders and birds at Mt Dale. This result concurs with an increasing body of evidence that show low levels of congruence in the diversity of different taxa (e.g., Prendergast ***et al.*** 1993; Abensperg-Traun ***et al.*** 1996; Oliver ***et al.*** 1998; Gunnar Jonsson & Jonsell 1999; Su ***et al.*** 2004). These findings have important ramifications for managers who are attempting to conserve jarrah forest biodiversity. A lack of congruence

among the diversity of relatively well-known taxa (e.g., birds, plants and ground spiders) shows that these taxa may not be adequate indicators of the biodiversity that is found in the jarrah forest. If this were the case, current strategies designed to protect biodiversity may not protect all taxa because these strategies are based predominantly on information gathered for well-known taxa.

The precautionary principle, an overarching principle to Ecologically Sustainable Forest Management (ESFM) (Calver 2003; Chapter 2), states that if there are threats of serious or irreversible environmental damage, the lack of full scientific certainty should not be used as a reason for postponing measures to prevent environmental degradation (Conservation and Land Management Act 2000). The different structural attributes found in forest disturbed by logging compared to unlogged forest suggests that logging disturbance may have serious impacts on not only the taxa assessed in this study, but also other relatively unknown taxa (e.g., saprotrophic fungi and soil cryptogams). As such, a larger inventory detailing the distribution, diversity and ecology of a wide range of taxa is needed before managers can confidently manage jarrah forests in an ecologically sustainable manner. The failure to use a range of indicators carries the risk that essential ecological processes may also be lost (Tilman 1997). The consequence of losing ecological processes is that even the taxa on which the conservation strategies were based are then at risk of long-term decline and loss.

The lack of congruence in species diversity among the three taxa is believed to occur because of the high levels of heterogeneity in species diversity among sites, and because each of the three taxa respond uniquely to logging and fire disturbance. With respect to species richness, the number of bird species

increased after gap logging, whereas the species richness of ground spiders and understorey plants decreased following logging (Chapter 5, 6 & 7). Furthermore, where there was a loss of understorey plant species after gap logging, there was replacement of ground spider species immediately after logging (some open ground hunters appeared to replace web weaving species) (Chapter 5 & 6). Therefore, although the three taxa responded similarly in that they were more severely affected by logging than fire, there was only small relationships in α and β diversity among taxa because each responded uniquely to these disturbances.

Lack of congruence in species diversity (α and β) has also been shown to be dependent on the spatial scale examined (Weaver 1994; Blair 1999). In my research, all three taxa had assemblages that varied across the 20 sites at Mt Dale (Chapter 4, 5, & 6). However, these highly heterogeneous patterns are not expected for some taxa at regional or landscape scales. Bird diversity is relatively homogeneous at larger spatial scales, with similar communities of species occurring throughout the jarrah forest (Abbott 1999). In contrast, understorey plants are known to be extremely heterogeneous across the jarrah forest landscape (Havel 1975a,b; Strelein 1988). Although not well studied, the composition of ground spider diversity is also expected to be highly heterogeneous across the jarrah forest landscape (Brennan 2002). These contrasting results have important implications with respect to reserve design. The current measures of placing 15% of jarrah forest in reserves could provide habitat that was used by all bird species that occur in jarrah forest. However, this reserve system is unlikely to conserve all species of understorey plants, or ground spiders. In other temperate forests, research has shown that conserving 20% of forest estate can be enough to sample all bird species, whereas up to 75%

of the forest is required to sample all understorey plant species (Saetersdal & Birks 1993). Given the importance of reserves for the conservation of viable populations of species affected by past and present logging practices, a greater awareness of the distribution, diversity and ecology of highly diverse taxa is required before it can be confidently concluded that the reserve system adequately, comprehensively and representatively conserves jarrah forest biodiversity.

8.4.1 Future Research

There are several benefits in measuring highly diverse taxa. With respect to maximising cost-efficiency, this research showed that the cost for sampling a site for each taxa were similar. The approximate costs (in Australian dollars) of the surveys that provided the data for this study were $2000 for understorey plants, $2100 for birds, and $2100 for ground spiders. These costs included field expenses and all salaries for sampling and processing, but do not consider the diversity found within each taxa. However, on a species-by-species basis, sampling for understorey plants ($8.84) was less than half the cost of sampling ground spiders ($35.00) and less than a quarter the cost of sampling birds ($53.84). Therefore, although some researchers argue that measuring plant and invertebrate diversity is time-consuming, expensive and often taxonomically difficult (Abensperg-traun *et al.* 1996), my research supports other studies that suggest surveying for invertebrate and vascular plant diversity is much less costly than surveying for vertebrates (e.g., Oliver & Beattie 1993; Horwitz *et al.* 1999; Lawton 2001).

Cost-efficiency is not the only consideration that must be made when assessing appropriate taxa. Understanding the spatial heterogeneity of biodiversity is an important consideration to the successful conservation of biodiversity in Western Australian forests (Wardell-Johnson & Horwitz 1996). Future research needs to focus on taxa that are highly heterogeneous across the landscape because these taxa may not be comprehensively, representatively or adequately sampled in the reserve system that currently exists in jarrah forest. As previously outlined, understorey plants (Havel 1975 a, b; Hopper 1992; Chapter 5) and ground spiders (Main 1987; Brennan 2002; Chapter 6) are examples of taxa that are specious and heterogeneous across small spatial scales in the jarrah forest. In contrast, birds are species-poor, are wide ranging and are capable of exploiting broad niches, and therefore are not particular useful taxa to investigate because most species are predicted to have populations found in the reserve system (Abbott 1999; Chapter 7). Any future research that assesses the success of the C.A.R. reserve system and ESFM in production forests, should make highly diverse taxa a focus because these are likely to contain species that are threatened by contemporary management practices.

8.4.2 Conclusion

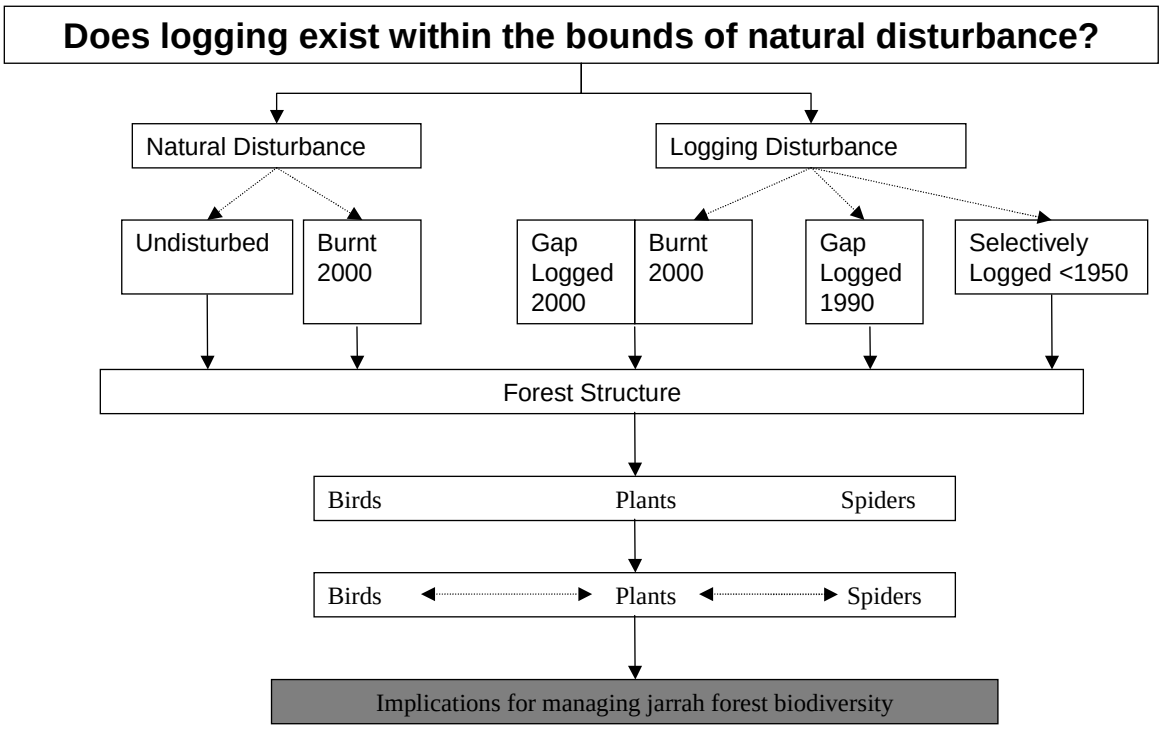
When all taxa are combined and treated as a measure of 'total biodiversity', differences in species richness and composition among treatments show that logging affects biodiversity for a longer period of time than fire affects biodiversity. Fires are managed on the assumption that biodiversity recovers in 5 -7 years. Differences in total biodiversity (both α and β) measured in forest gap logged in 1990 compared to undisturbed forest shows that biodiversity had not fully recovered 10 years after gap logging disturbance. Furthermore, although

the total number of species was similar between forest selectively logged prior to 1950 and undisturbed forest, there were significant differences in the assemblages of species within these two treatments. In fact, the composition of species in forest selectively logged prior to 1950 was more similar to communities in forest gap logged in 1990 than undisturbed forest. Given selective logging prior to 1950 was less intensive than gap logging, these results imply that contemporary logging practices in jarrah forest may affect biodiversity for at least 50 years, which is a much greater time than fire affects biodiversity.

The second major finding of this chapter was the lack of congruence in the diversity of understorey plants, ground spiders and birds among sites at Mt Dale. This has important implications for managers that are attempting to conserve jarrah forest biodiversity. This result shows that conservation measures cannot simply rely on information of a few well-known taxonomic groups to conserve jarrah forest biodiversity. Instead, managers need to incorporate information from a wide range of taxa into management practices. By doing this, managers can more confidently conclude that jarrah forest biodiversity is not being detrimentally impacted by logging disturbance.

CHAPTER 9

Implications for Management



Chapter 9 Conserving Jarrah Forest Biodiversity

The objective of this thesis was to assess whether logging disturbance emulates fire disturbance in jarrah forest. This is because a central axiom of Ecologically Sustainable Forest Management is that any manipulation of a forest ecosystem should emulate the 'natural' disturbance patterns of the region that existed prior to 'industrial' manipulation, such as logging (Burrows *et al.* 2002a). ESFM is based on the assumption that forest biodiversity has evolved with endogenous disturbance regimes (disturbances that are part of the natural system, and part of autogenic development), and will be better able to cope if exogenous disturbance (disturbances that are introduced to the system, and part of allogenic development) remains within natural bounds of spatial and temporal severity (Franklin 1989; Gillis 1990; Hansen *et al.* 1991; Haila *et al.* 1994; Attiwill 1997; Stork *et al.* 1997; Peterken 1999; Mitchell *et al.* 2002). Exogenous disturbances such as logging may result in the loss of biodiversity if this action exists beyond the boundaries determined by previous natural disturbance regimes (Norton 1996; Attiwill 1997; Lindenmayer & Recher 1998).

Four indicators of biodiversity (forest structure, understorey plants, ground dwelling spiders and birds) showed that there were differences in the way biodiversity was affected by these disturbances at Mt Dale (Chapters 4 - 8). These indicators showed there were differences in the biodiversity in forest disturbed by gap logging and fire in 2000 compared to unlogged forest that was disturbed by fire in 2000. Moreover, they showed there were small, but significant, differences in the biodiversity found in unlogged forest compared to forest gap logged in 1990 and, to a lesser extent, forest selectively logged prior to 1950. These results show that logging disturbance influences biodiversity for a

longer period of time (e.g., for at least 10 years after gap logging and 50 years after selectively logging) than fire disturbance (maximum of 7 years). Therefore, the immediate impact, and subsequent recovery, of biodiversity from logging disturbance cannot be considered analogous to fire disturbance in the jarrah forest.

The idea that logging emulates fire disturbance was assessed because the current axiom of Ecologically Sustainable Forest Management (ESFM) is that any manipulation (e.g., logging) of a forest ecosystem should emulate the 'natural' disturbance regime of the region (Franklin 1989; Gillis 1990; Hansen ***et al.*** 1991; Haila ***et al.*** 1994; Attiwill 1994; Stork ***et al.*** 1997; Hunter 1999; Peterken 1999; Burrows ***et al.*** 2002a). In jarrah forest, no natural disturbance regimes have the same severity, spatial pattern or frequency as the disturbance caused by contemporary silvicultural techniques. This is because gap logging aims to regenerate the stand from underlying coppice and therefore is a stand replacing disturbance that causes the mortality of all trees in a coupe except habitat trees. The multi-age structure of trees in unlogged jarrah forest indicates that naturally occurring stand replacing disturbances such large fires, hurricanes (blow-downs), floods and storms occurred very infrequently in jarrah forest (Abbott & Loneragan 1986). The major natural disturbance regimes in the jarrah forest were low to moderate intensity fires that do not initiate the regeneration of stands of jarrah trees (Burrows ***et al.*** 2002a). Contemporary silviculture by gap logging disturbance therefore does not lie within the intensity of disturbance regimes that occur naturally in jarrah forest.

Logging disturbance is also managed at different spatial and temporal scales than fire disturbance in jarrah forest. With respect to spatial scale, patches disturbed by gap-logging are a maximum of 10 ha in size (Department of Conservation and Land Management 1995). In contrast, prescribed fires are managed over a much larger areas (100 - 1000s ha), the boundaries of which are determined by artificial (e.g., roads and dams) and natural barriers (e.g., granite outcrops, boundaries of past fires) (Department of Conservation and Land Management 1997). With respect to the frequency of these two disturbance events, fires are planned every 5 – 7 years in most forest blocks, where as logging is expected to every 100 – 220 years, with thinning potentially occurring every 20 years (Ferguson *et al.* 2003; Conservation Commission of Western Australia 2004). Logging disturbance therefore does not exist within any of the boundaries determined by the severity, spatial pattern or frequency of the predominant natural disturbance regime in jarrah forest. As such, logging needs to be recognised as an exogenous disturbance that could be capable of causing changes to biodiversity of the jarrah forest (Norton 1996; Attiwill 1997; Lindenmayer & Recher 1998).

The degree to which jarrah forest biodiversity is altered by logging, and the length of time biodiversity is altered, is only beginning to be understood (see Craig 1999; Abbott *et al.* 2002; Burrows *et al.* 2002b; Strehlow *et al.* 2002; Abbott *et al.* 2003; this research). At Mt Dale, similar richness of taxa was recorded in forest disturbed by gap logging, forest disturbed by selective logging and unlogged forest, which suggests that biodiversity is relatively resilient to logging disturbance. However, there were small, but significant, differences in the diversity of particular guilds in forest gap logged in 1990 compared to

unlogged forest. This implies that some resources required by these guilds take at least 10 years to recover following disturbance by gap logging. With respect to understorey plants, the low number of viable seeds produced by some regenerating species has led to speculation that it could take a considerable period of time for seed banks, and therefore plant communities, to return to pre-logging conditions (Burrows *et al.* 2002b). Similarly, lower numbers of web-building spiders in forest gap logged in 1990 compared to unlogged forest suggest that the microhabitats required some of these species are not available in patches recently disturbed by gap logging. In other temperate forests, these microhabitats are often related to attributes that can take up to 30 years to return after logging (Huhta 1971; Heliovaara & Vaisanen 1984; Atlegrim & Sjoberg 1995). The length of time it may take these resources to return to patches that are disturbed by gap logging highlights the need to appropriately manage stands that are not affected by this disturbance.

9.1 Conserving Biodiversity: Protecting Unlogged Forest

The protection of unlogged forest from logging disturbance is the most recent attempt by the Western Australian State Government to conserve the aesthetic, cultural and biodiversity values of forest in southwest Australia (Government of Western Australia 2001). It is estimated that 10 % of the jarrah forest is unlogged ('old-growth') (Conservation Commission of Western Australia 2004). In the region where my research was undertaken, the 'Jarrah Northwest' (see Fig 2.4), 1.2 % (8 095 ha) of the forest that existed prior to European arrival is unlogged (Conservation Commission of Western Australia 2004). These unlogged patches are found in C.A.R. formal and informal reserves (as described in Section 2.5), and are 'relictual' forest landscapes if the Landscape-

Continuum Model is used to interpret the current extent of these reserves (Fig. 9.1). When the previously logged forest that also exists within the C.A.R. reserve system recovers from logging disturbance, and contain the same resources as unlogged forest, 15% of the forest will be managed under natural disturbance regimes (Fig. 9.2). These reserves represent a 'heavily fragmented' landscape using the Landscape-Continuum Model (Fig. 9.1). At Mt Dale, unlogged patches contained more species per hectare than forest that had been disturbed by gap logging, and contained small, but significant, differences in the composition of species with gap logged forest, and to a lesser degree, forest selectively logged prior to 1950. As such, my research provides evidence that the conservation of remnant unlogged forest in the jarrah northwest is an appropriate precautionary measure to conserve biodiversity in this region.

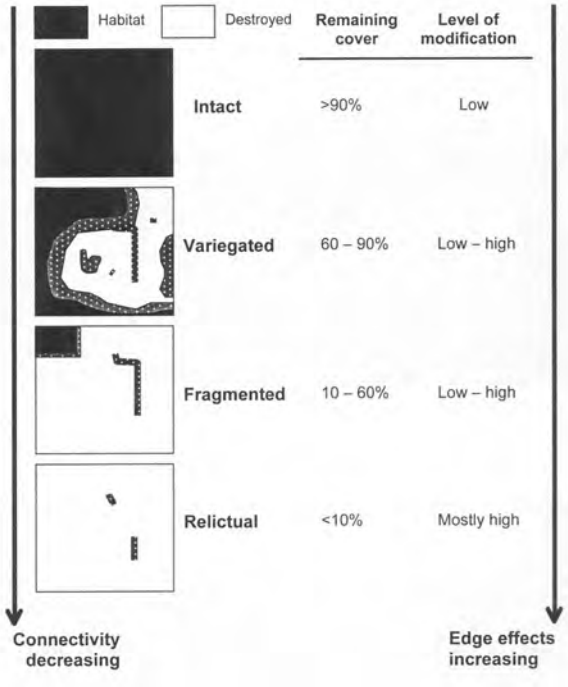


Fig. 9.1 Four states of landscape alteration defined by the degree of habitat destruction in the landscape continuum model (modified from McIntyre & Hobbs 1999; Lindenmayer & Franklin 2002). Unlogged forests have gone from an intact landscape prior to European arrival to a relictual landscape. All other forest have been cleared or selectively logged.

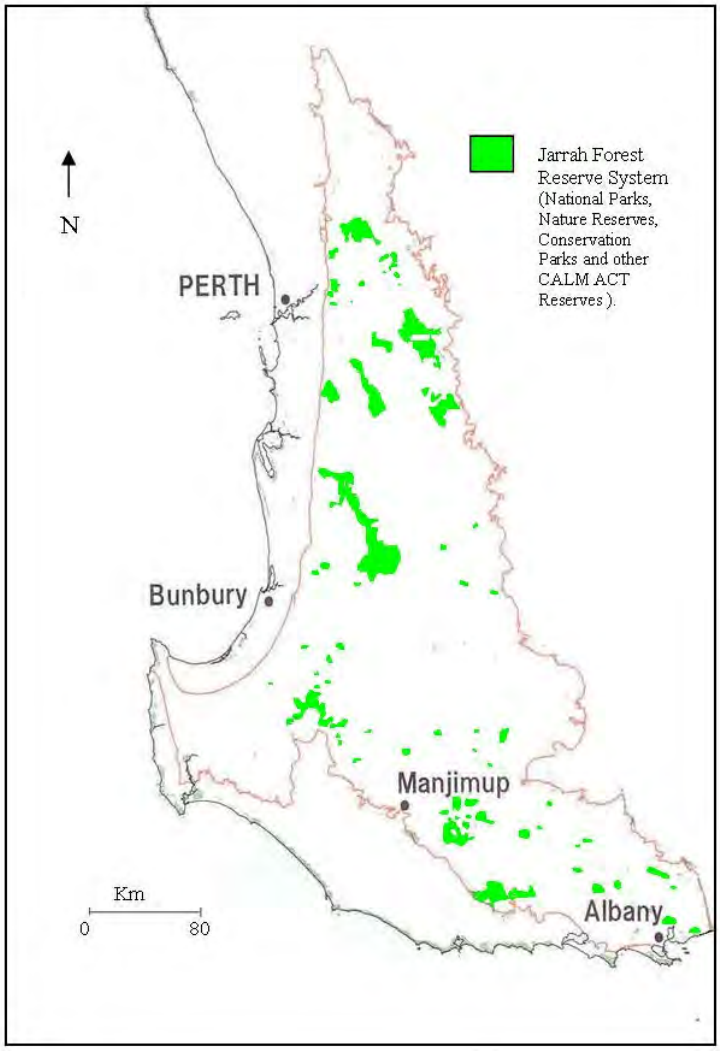


Fig. 9.2 The jarrah forest reserve system (modified from Department of Conservation and Land Management 2005).

Protecting remnants of unlogged forest as a basis to conserve forest biodiversity may not be an appropriate strategy to conserve biodiversity in jarrah forest. This is because patches of unlogged forest may not have been logged because they were less productive with respect to timber yields than other, more productive, stands. If this were the case, the biodiversity now found in unlogged forest may not be representative of biodiversity that is found throughout the jarrah forest. At Mt Dale, comparisons between the size of stumps in logged stands with the basal area of live trees in unlogged stands indicates that unlogged and logged stands

had similar size structure of trees prior to logging. As more species were recorded per site in unlogged forest than forest disturbed by logging, my results indicate that there could be a loss of a small number of species if the remaining unlogged forest were disturbed by gap logging. Although my study was limited to a small spatial scale, and should not be extrapolated across the jarrah landscape, this research provides evidence that the protection of unlogged forest from logging is an appropriate measure to conserve jarrah forest biodiversity. However, future research in other unlogged patches is required to assess whether protecting unlogged forest from logging will conserve biodiversity at a landscape scale.

In other temperate forests, unlogged patches have been shown to be important reservoirs of biodiversity that are detrimentally impacted by logging (e.g., Heliovaara & Vaisanen 1984; Zielinski & Gellman 1998; see review by Burgman 1996). Different organisms have been shown to benefit from the protection of unlogged patches. The first are organisms that have relatively high dispersal capability but depend on structural resources found in mature forests to forage, breed or roost (Loyn 1998; Rolstad *et al.* 2002). Within a forest landscape that has become highly fragmented by logging, these species utilise resources within unlogged forest, and re-colonise stands that are disturbed by logging when the resources they require recover from this disturbance. In other temperate forests, these types of organism includes owls (Doak 1989; Lattin & Moldenke 1992), possums (Lindenmayer *et al.* 1991; Goldingay & Possingham 1996), and bats (Kutt 1995; Zielinski & Gellman 1998). If metapopulation dynamics apply to these species, populations in unlogged forest become 'source' populations to patches that have been altered, and recovered, from logging

disturbance (Mladenoff *et al.* 1993). At Mt Dale, some species of bird may be utilising unlogged forest as breeding habitat but forage in recently disturbed forest (as shown by Craig 1999). In theory, these types of organisms should be able to persist if enough forest is left unlogged at any one time to support viable populations, and if resources are able to develop and be restored in patches disturbed by logging (Rolstad *et al.* 2002).

The second group of species that depend on resources in unlogged forest are those organisms that are negatively affected by logging, but have poor dispersal ability. In jarrah forest, many organisms including spiders, isopods, vascular and non-vascular plants species have low vagility (Hopper *et al.* 1996). At Mt Dale, regenerating vascular plants and web-building spiders were found to be negatively affected by contemporary silviculture by gap logging, and could therefore represent this type of organism. If these taxa are negatively affected by logging, their ongoing persistence depend on the presence of appropriate stand structures continuously over long time periods, where care should be taken to preserve these stands within a matrix of production forest that are managed to enhance the dispersal of these species (Rolstad *et al.* 2002).

The fragmentation of once continuous populations could result in the loss of biodiversity. Research in other temperate forests has shown that the subdivision and isolation of unlogged patches has increased the amount of edge habitat in these patches resulting in changes in relationships among taxa (e.g., inter-specific and intra-specific competition, and predation). Such changes can have a profound effect on all levels of biodiversity found in unlogged forest (MacArthur & Wilson 1967; Diamond 1975; Forman 1995; Komonen 2001). Severe

fragmentation can cause losses at genetic, population/species and community/ecosystem levels of biodiversity (e.g., Saunders *et al.* 1993; Goldingay & Possingham 1995; Redford & Richter 1999; Watson 2004). Moreover, fragmentation can lead to major abiotic alterations including changes to hydrological regimes, mineral nutrient cycles, radiation balance, wind patterns and soil movement (Saunders *et al.* 1993; Hobbs 1993). Ecologists have long recognised that the reliance on a system of reserves designed to protect the remaining unlogged forest will not conserve all forest species (Recher 1985; Norton & Kirkpatrick 1995; Hobbs 1996; Lindenmayer and Franklin 1997; Mitchell & Craig 2000). Given that the majority of the forests that are not in reserves are production forests that are managed for timber and pulp (e.g., 58 % of the 502 950 ha of publicly owned forest in the jarrah northwest region can be logged under the new forest management plan) (Conservation Commission of Western Australia 2004), the successful conservation of jarrah forest biodiversity requires appropriately managing the production forests in addition to protecting remaining unlogged forests. Production forests can be managed to have a 'connectivity' function (enhancing dispersal of some organisms between unlogged forests); a 'lifeboating' function (allowing organisms to persist in areas from areas in which they would otherwise be eliminated); and a 'habitat heterogeneity' function (providing the variation in habitat conditions required by some taxa) (Lindenmayer & Franklin 2003).

9.2 Conserving Biodiversity: Managing Production Forests

9.2.1 Biodiversity in 'Mature' Forests

The new forest management plan for jarrah forest (Forest Management Plan 2004-2013) has implemented significant changes with regards to management of mature stands in production forests. In jarrah forest, 'mature forest' is defined as

"forest that meets the JANIS (1997) definition of old growth, or forest that has a mature overstorey (height and cover) and has a preponderance of trees in the larger diameter classes" (Burrows *et al.* 2002b p. 25).

Fauna Habitat Zones (FHZs) are selected with regards to mature habitat elements, and will be conserved to provide a source population of mammals that can recolonise nearby areas that have been disturbed by logging. Research at Mt Dale showed that forest selectively logged prior to 1950 can be considered to be mature forest. These forests have the same height and cover as trees in unlogged trees, and a significant proportion of the trees in these forests are larger than 1m d.o.b (Chapter 4). My research shows that if these mature forests consisted of forest that were selectively logged 50+ years ago, FHZs should also conserve some guilds of non-fauna species (e.g., web-weaving spiders and regenerating nanophanerophytes) that are impacted by contemporary logging practices.

FHZs zones currently represent approximately 55 000 ha (~ 6% of the production forest, and less than 2% of pre-European jarrah forest) (Conservation Commission of Western Australia 2004). These zones are to be distributed as 200ha patches within forest blocks that will be logged. FHZs are not permanent conservation reserves, and can be rotated with alternative areas of regenerating forest. The period of time between the implementation of FHZs and the time

they are logged is unspecified. Given that most of the production forest will be logged in the next 220 years (Ferguson *et al.* 2003), the conservation of biodiversity that require mature forest in production areas will ultimately depend on forest that has been disturbed by contemporary logging practices.

9.2.2 Biodiversity in forest disturbed by contemporary logging

Several new prescriptions designed to conserve components of diversity in logging coupes have been implemented in the new forest management plant (Conservation Commission of Western Australia 2004). A key feature of the new practices is enhanced provision for conservation of biodiversity through increased retention of habitat, limits on culling and greater controls on disturbance to the subdominant trees, understorey and the soil (Stoneman *et al.* 2005). The new prescriptions include (from Conservation Commission of Western Australia 2004):

- Increasing the retention of habitat trees to 5 ha^{-1} (from 4 ha^{-1}) and for potential habitat trees to 6-8 ha^{-1} (from 6 ha^{-1});
- Retaining all hollow logs with a pipe > 10cm diameter, and > 3 m in length, as opposed to conserving one large log or stump ha^{-1};
- That pushing over understorey species to promote regeneration of jarrah will only be undertaken in areas where there is clear evidence of the past presence of jarrah, such as old stumps. These push down treatments of understorey are to focus on groups, or 'clumps', of species such as bull banksia (***Banksia grandis***) and sheoak (***Allocasuarina fraserianal***) that are impeding regeneration establishment. Approximately 20% of these clumps should be retained as small clumps (0.02 ha) or as scattered individuals.
- Balga (***Xanthorrhoea preissii***) 'thickets' are to be preserved.
- Mature individuals of species of balga, woody pear (***Xylomelum occidentale***), river banksia (***Banksia verticillata***), snottygobble (***Persoonia longifoliaI***), Western Australian peppermint (***Agonis flexuosa***), and Western Australian christmas tree (***Nuytsia floribunda***) will be retained.
- Disturbance of the top soil is to be avoided where 'reasonable' and 'practicable'.
- Reducing excess coarse woody debris waste by stopping culling in areas that have stand basal area of greater than 17m^2 ha^{-1} after regeneration release.

With respect to the three taxa investigated at Mt Dale, these changes are likely to increase the resources required by species detrimentally impact by gap logging. For example, reductions to the disturbance of the top soil are likely to enable

more regenerating plant species to exist in these coupes. The retention of clumps of subdominant trees (e.g., *B. grandis* and *A. fraseriana*), and the reduction in the amount of vegetation that is pushed down, will also increase foraging and nesting for birds, and can be expected to provide some microhabitats required by ground spiders to persist after logging. These new prescriptions effectively reduce the amount of forest that is disturbed by machinery, which increases the retention of breeding, roosting and foraging habitat utilised by organisms that exist in unlogged jarrah forest.

Although the new silviculture prescriptions are a positive action with respect to conserving biodiversity in logged forest, these prescriptions do not reduce the potential gap logging has to homogenise tree size and age (see Burrows *et al.* 2002a). This homogenisation is predicted because current silvicultural guidelines are still primarily designed to produce millable timber. For example, there are no clearly defined objectives for non-wood related products in calculating the rotation length in jarrah forest (Ferguson *et al.* 2003). Instead, current rotation periods are calculated on a sustained yield basis, which is designed to produce smaller (in terms of basal area rather than height), more even aged stands than naturally occur (Burrows *et al.* 2002a). As previously outlined, approximately 58% of the jarrah northwest region is is planned to be logged using contemporary techniques in the next rotation (100 - 220 years) (Conservation Commission of Western Australia 2004). As forest structural diversity is likely to be related to many aspects of forest biodiversity (MacArthur & MacArthur 1961; Chapter 4), the homogenisation of large areas of jarrah forest could influence the biodiversity across the jarrah forest landscape. Populations of species that require resources within mature/over-mature forests

(i.e., those stands that contain unlogged structural attributes such as multi-age structure) may only be found in CAR formal and informal reserves, and therefore suffer from problems associated with fragmentation.

9.3 Using 'Natural' Disturbance Regimes to Manage the Jarrah Forest

This research was undertaken because ESFM has a central axiom that any manipulation (e.g., logging) of a forest ecosystem should emulate the 'natural' disturbance regime of the region. In other temperate forests, many new silvicultural systems have emerged during recent decades based on the assumption that diversity patterns and ecological processes are more likely to persist if disturbances occurring through management mimic the patterns and processes of natural disturbances (e.g., Franklin 1989; Gillis 1990; Mitchell *et al.* 2002). I believe there are economic and ecological reasons why it is inappropriate to attempt to use natural disturbance as a paradigm to manage jarrah forest. With respect to economics, a logging regime that attempts to re-create this age-structure would have to be based on extremely long rotations because of the longevity and the slow growth of jarrah trees (O'Connell & Mendham 2004). This change in silviculture would jeopardise the Western Australian Regional Forest Agreement (Commonwealth of Australia and Government of Western Australia 2004) and the current Forest Management Plan (Conservation Commission of Western Australia 2004) because it would substantially reduce the economic viability of the timber industry in this state.

The second reason that contemporary silviculture should not attempt to emulate natural disturbance is that jarrah forests have not existed under the same 'natural'

disturbance regime for at least 50 millennia (Ehrlich 1996). Large changes in climate occurred as recently as 7000 years ago that resulted in the expansion and contraction of the forests of Western Australia (Chapter 2; Semeniuk 1995). Aboriginal regularly disturbed forests using fire practices that were very different to the fire regimes that result from lightening strike. More recently Europeans have substantially influenced the structure of jarrah forests and associated biota that occur within it by introducing new organisms into the ecosystem, and new disturbance regimes (Watson 2005). The use of natural disturbance regimes is based on the notion that past natural conditions and processes provide the context and guidance for managing ecological systems today (Landres ***et al.*** 1999). However, if the organisms that now exist 'naturally' in the jarrah forest are in part the product of past disturbance regimes that have consistently changed through time, then it is impossible to select a single 'natural disturbance regime' as the basis for silvicultural guidelines. Contemporary managers should recognise that effects associated with contemporary logging disturbance occur in addition to changes that have already resulted from past disturbance regimes (Norton 1996). Instead of attempting to manage forests to occur within the boundaries of natural disturbance, jarrah forests are probably better managed with specific objectives relating to biological conservation. The extent to which the forests have been disturbed by logging means that one objective should be to conserve communities that exist in mature/overmature forest by promoting the ongoing existence of these habitats.

An important consideration underpinning silviculture in wood production forests should be the recognition that the same harvesting system does not necessarily have to be applied to all areas targeted for logging (Lindenmayer & Recher

1998). In other temperate forests, Variable Retention Harvest Systems (VRHS) have been utilised to ensure the ecology and economic sustainability of forest ecosystems (Lindenmayer & Franklin 2003). Implicit in VRHS is an acceptance that some of the productive capacity and economic value of the stand will be devoted to the maintenance of biodiversity, rather than simply maximising the regeneration and growth of commercial tree species (Sullivan *et al.* 2001; Franklin *et al.* 1997). Given the uncertainty that surrounds the future biodiversity within homogeneous (even aged) stands, a VRHS approach in jarrah forest would result in a proportion of logged patches being allowed to recover unlogged structural characteristics (i.e., being on much longer rotations than 200 years). No reduction in the current availability of sawlogs would exist, and therefore the current Western Australian Regenional Forest Agreement (Commonwealth of Australia and Government of Western Australia 1999) or Forest Management Plan (Conservation Commission of Western Australia 2004) would not be jeopardised. However, future agreements would recognise that reductions in sawlog availability need to occur as some patches would not be logged for timber and pulp for many years.

Models like VRHS offer greater flexibility in relation to the future conservation of biodiversity at patch and landscape scales than are found in contemporary silvicultural guidelines. At an operational scale, species that require mature/overmature jarrah forest will return to logged patches when they contain the structural attributes found in unlogged forest. At a landscape level, patches managed on longer rotations could be designed to buffer and connect overmature/mature forest in reserves and production forests. These logging operations can be changed to recognise the natural variability that occurs within

the jarrah forest system, as abiotic factors (e.g., precipitation, soil and site properties) may control rate, scale and severity of disturbance in some areas (Kramer *et al.* 2001). The implementation of a system like VRHS would allow for the future revariegation the jarrah forest, and could be managed to alleviate against many of the problems associated with fragmentation of unlogged forest.

9.4 Using Appropriate Measures of Biodiversity in Jarrah Forest

The results of this thesis show that species richness is not a good predictor of biodiversity in jarrah forest. This is because the recovery of species richness does not indicate the resilience of communities following this disturbance. Marked differences in the composition of species before and after logging were noted when numbers of species appeared equivalent (e.g, regenerating phanerophytes and air foraging birds) (Chapter 5 & Chapter 7). Furthermore, univariate analysis of species richness often did not have the power to detect differences, and therefore there was a significant risk of making a Type II error (i.e., stating there was no difference between treatments when there actually was a difference between treatments). One aspect of a precautionary approach to Ecologically Sustainable Forest Management is that managers must try and avoid Type II errors because the outcome could be a loss of biodiversity (see Calver 2003). Moreover, measuring species richness is nonsensical because it does not allow managers to identify which species are being negatively impacted by logging disturbance. Future assessments of the effects of logging should monitor assemblages to species that occur in disturbed areas, rather than examining whether species numbers return to pre-disturbance levels.

The composition of a suite of indicators that apply to different ecological features need to be used understand the impacts of logging on biodiversity. The lack of surrogacy among the three taxa investigated at Mt Dale shows that any research that assesses logging disturbance is likely to be taxa specific. Future research therefore needs to focus on a range of taxa that have life history traits likely to be impacted by logging. These organisms could include some of guilds recognised in this study (e.g., web building spiders and regenerating phanerophytes), as well as other taxa that require structural attributes that have shown to be affected by contemporary logging practices (e.g., organisms that utilise old trees and logs).

9.5 Conclusion

This thesis has shown that logging disturbance does not emulate the predominant natural disturbance regime in jarrah forest (fire). This has major ramifications for forest managers. As more species occurred in unlogged forest than surrounding production forest at Mt Dale, the protection of unlogged remnants is seen to be an appropriate strategy to conserve forest biodiversity. The similar biodiversity found in forest selectively logged prior to 1950 and unlogged forest means that mature stands of trees in the production forest could be managed to add to the conservation of unlogged forest. The fragmented areas of unlogged forest can be connected, and buffered from external disturbances, using these mature forests. This thesis has also shown that gap logging could have ongoing affects on biophysical attributes, and that forest managers must start to consider new strategies to alleviate any possible negative impacts.

The jarrah forest is recognised as highly biodiverse region of the world. ESFM within jarrah forest requires all new disturbances (logging, mining, road

building) to be integrated within a landscape dedicated to biological conservation (as well as other purposes including water catchment protection and recreation). The development of strategic forest management approaches and silvicultural techniques to maintain the spectrum of forest compositions and structures at different scales in the landscape is required to conserve the biodiversity found in jarrah forest. Unlogged and logged jarrah forest must be managed in harmony to sustain a timber industry as well as protect species that require resources found in mature/over-mature stands.

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Appendix 1

Changes in the way people perceive the biophysical condition of the jarrah forest. pgs 225-231 in ***Proceedings of the 6TH National Conference of the Australian Forest History Society***. Calver, M. ***et al.*** (eds). Millpress, Rotterdam.

Not included here

Appendix 2 The latitude and longitude of each of site.

Treatment	Site Name	Latitude	Longitude
Undisturbed	U1	32^0 07' 33.2" S	116^0 17' 42.5" E
	U2	32^0 06' 49.3" S	116^0 17' 23.1" E
	U3	32^0 05' 23.5" S	116^0 17' 45.9" E
	U4	32^0 06' 03.9" S	116^0 17' 14.8" E
Burnt 2000	B1	32^0 07' 09.5" S	116^0 17' 28.3" E
	B2	32^0 07' 19.1" S	116^0 17' 33.4" E
	B3	32^0 06' 25.5" S	116^0 17' 05.4" E
	B4	32^0 07' 25.4" S	116^0 17' 36.3" E
Gap logged / Burnt 2000	GLB1	32^0 06' 22.4" S	116^0 16' 53.3" E
	GLB2	32^0 06' 14.3" S	116^0 16' 35.9" E
	GLB3	32^0 06' 43.0" S	116^0 17' 15.1" E
	GLB4	32^0 06' 06.8" S	116^0 16' 17.8" E
Gap logged in 1990	GL90 1	32^0 05' 38.5" S	116^0 16' 49.3" E
	GL90 2	32^0 05' 18.5" S	116^0 16' 51.5" E
	GL90 3	32^0 05' 57.1" S	116^0 17' 06.2"E
	GL90 4	32^0 05' 13.0" S	116^0 17' 33.6" E
Selectively logged prior to 1950	SL50 1	32^0 06' 10.0" S	166^0 17' 10.1" E
	SL50 2	32^0 05' 30.4" S	116^0 16' 40.4" E
	SL50 3	32^0 05' 00.1" S	116^0 16' 59.3" E
	SL50 4	32^0 05' 15.9" S	116^0 16' 42.4" E

Appendix 3 The understorey plant species recorded at Mt Dale. Plants are categorised into guilds; 'OS Na' - Obligate seeding nanophanorephytes, 'OS He' - Obligate seeding hemicryptophytes, 'RE Na' - Regenerating nanophanerophytes, 'RE He'- Regenerating hemicryptophytes, 'RE Cry'- Regenerating cryptophytes. 'No Gu'refers to the taxa were not classified because they could not be identified to species (n = 25) or because of a lack of biological information (n = 19).

FAMILY	Species	OS Na	OS He	RE Na	RE He	RE Cry	No Gu
AMARANTHACEAE	*Ptilotus esquamatus* (Benth.) F.Muell.	1					
	Ptilotus polystachyus (Gaudich.) F.Muell.	1					
	Ptilotus manglesii (Lindl.) F.Muell.		1				
ANTHERICACEAE	*Anthericaceae* sp. 2						1
	Chamaescilla corymbosa (R.Br.) Benth.					1	
	Thysanotus patersonii R.Br.					1	
	Laxmannia squarrosa Lindl.	1					
	Laxmannia sp. 1	1					
	Thysanotus sparteus R.Br.					1	
	Thysanotus sp. 1					1	
	Thysanotus sp. 2					1	
	Thysanotus sp. 3.					1	
APIACEAE	*Pentapeltis peltigera* (Hook.) Bunge				1		
	Xanthosia atkinsoniana F.Muell.			1			
ASTERACEAE	*Lagenofera huegelii* Lam.		1				
	Trichocline spathulata (DC.) J.H.Willis		1				
BORAGANACEAE	*Halgania corymbosa* Lindl.						1
	Allocasuarina fraseriana (Miq.) L.A.S.Johnson			1			
CASUARINACEAE							
COLCHICACEAE	*Burchardia multiflora* Lindl.					1	
	Burchardia umbellata R.Br.					1	
CYPERACEAE	*Lepidosperma pubisquameum* Steud.					1	
	Lepidosperma tenue Benth.					1	
	Lepidosperma tetraquetrum Nees					1	
	Lepidosperma sp. 1					1	
	Lepidosperma sp. 2					1	
	Lepidosperma sp. 3					1	
	Lepidosperma. sp. 4					1	
	Lepidosperma sp. 5					1	
	Mesomelaena tetragona (R.Br.) Benth.					1	
	Mesomelaena sp. 1					1	
	Mesomelaena sp. 2					1	
	Tetraria octandra (Nees) Kuk.						1
DASYPOGONCEAE	*Kingia australis* R.Br			1			
	Lomandra integra T.Macfarlane					1	
	Lomandra micrantha (Endl.) Ewart					1	
	Lomandra preissii (Endl.) Ewart					1	
	Lomandra sp.1					1	
	Lomandra caespitosa (Benth.) Ewart						1
DENNSTAEDIACEAE	*Pteridium esculentum* (G.Forst.) Cockayne						1
DILLENACEAE	*Hibbertia amplexicaulis* Steud.			1			
	Hibbertia cuneiformis (Labill.) Sm.			1			

FAMILY	Species	OS Na	OS He	RE Na	RE He	RE Cry	No Gu
	Hibbertia serrata Hotchk.			1			
	Hibbertia commutata Steud.			1			
	Hibbertia hypericoides (DC.) Benth.			1			
	Hibbertia pachyrrhiza Steud.			1			
DROSERACEAE	*Drosera erythrorhiza* Lindl.				1		
	Drosera gigantea Lindl.				1		
	Drosera menziesii DC.				1		
	Drosera pallida Lindl.				1		
	Drosera stolonifera Endl.				1		
	Drosera sp. 1				1		
	Drosera sp. 2				1		
EPACRIDACEAE	*Astroloma ciliatum* (Lindl.) Druce			1			
	Astroloma glaucescens Sond.			1			
	Astroloma pallidum R.Br.			1			
	Astroloma *sp.1*						1
	Astroloma sp. 2						1
	Epacridaceae sp. 1						1
	Leucopogon oxycedrus Sond.			1			
	Leucopogon propinquus R.Br.			1			
	Leucopogon pulchellus Sond.			1			
	Leucopogon verticillatus R.Br.			1			
	Leucopogon sp.1			1			
	Styphelia tenuiflora Lindl.	1					
EUPHORBIACEAE	*Phyllanthus calycinus* Labill.			1			
	Stachystemon vermicularis Planch.						1
FABACEAE	*Bossiaea ornata* (Lindl.) Benth.			1			
	Daviesia cordata Sm.			1			
	Daviesia decurrens Meisn.			1			
	Daviesia polyphylla Benth.			1			
	Daviesia preissii Meisn.			1			
	Daviesia flexuosa Benth.			1			
	Eutaxia dillwynioides Meisn.						1
	Gastrolobium calycinum Benth.	1					
	Gompholobium knightianum Lindl.	1					
	Gompholobium marginatum R.Br.	1					
	Gompholobium preissii Meisn.	1					
	Hovea chorizemifolia (Sweet) DC.			1			
	Hovea elliptica (Sm.) DC.			1			
	Kennedia coccinea Vent.	1					
	Labichea punctata Benth.			1			
	Sphaerolobium medium R.Br.				1		
GOODENIACEAE	*Dampiera alata* Lindl.						1
	Dampiera linearis R.Br.						1
	Dampiera trigona de Vriese						1
	Lechenaultia biloba Lindl.			1			
	Scaevola calliptera Benth.						1
	Scaevola pilosa Benth.						1
HAEMODORACEAE	*Anigozanthos manglesii* D.Don				1		
	Conostylis caricina Lindl.				1		
	Conostylis serrulata R.Br.				1		
	Conostylis sp.1				1		

FAMILY	Species	OS Na	OS He	RE Na	RE He	RE Cry	No Gu
	Haemodorum laxum R.Br.					1	
	Haemodorum simulans F.Muell.					1	
	Haemodorum sp. 1					1	
	Haemodorum sp. 2					1	
	Haemodorum sp. 3					1	
	Haemodorum sp. 4					1	
	Haemodorum sp. 5					1	
IRIDACEAE	*Orthrosanthos laxus*				1		
	Patersonia pygmaea Lindl.				1		
	Patersonia occidentalis R.Br.				1		
LAMIACEAE	*Hemiandra pungens* R.Br.			1			
LAURACEAE	*Cassytha racemosa* Nees						1
LINDSAEACEAE	*Lindsaea linearis* Sw.						1
LORANTHACEAE	*Nuytsia floribunda* (Labill.) Fenzl						1
MIMOSACEAE	*Acacia alata* R.Br.	1					
	Acacia applanata Maslin	1					
	Acacia lasiocarpa Benth.	1					
	Acacia pulchella R.Br.	1					
	Acacia stenoptera Benth.	1					
	Acacia urophylla Lindl.	1					
	Acacia sp.1	1					
	Acacia sp. 2	1					
MYRTACEAE	*Agonis linearifolia* (DC.) Sweet			1			
	Astartea fascicularis (Labill.) DC.			1			
	Baeckea camphorosmae			1			
	Calothamnus quadrifidus R.Br.			1			
	Corymbia calophylla (Lindl.) K.D.Hill & L.A.S.Johnson			1			
	Eucalyptus marginata Sm.			1			
	Hypocalymma angustifolium (Endl.) Schauer			1			
	Melaleuca incana R.Br.			1			
	Myrtaceae sp. 1						1
	Myrtaceae sp. 2						1
ORCHIDACEAE	*Burndettia nigricans*					1	
	Caladenia flava R.Br.					1	
	Caladenia deformis R.Br.					1	
	Caladenia longicauda Lindl.					1	
	Elythranthera brunonis (Endl.) A.S.George					1	
	Pterostylis aspera D.L.Jones & M.A.Clem.					1	
	Pterostylis barbata Lindl.					1	
	Pterostylis nana R.Br.					1	
	Pterostylis recurva Benth.					1	
	Pterostylis sp. 1					1	
	Thelymitra crinita Lindl.					1	
POACEAE	*Neurachne alopecuroidea* R.Br.		1				
	Tetrarrhena laevis R.Br.						1
	Themeda triandra Forssk.						1
PHORMIACEAE	*Dianella revoluta* R.Br.				1		

FAMILY	Species	OS Na	OS He	RE Na	RE He	RE Cry	No Gu
PITTOSPORACEAE	*Pronaya fraseri* (Hook.) E.M.Benn.					1	
	Sollya heterophylla Lindl.					1	
POLYGALACEAE	*Comesperma virgatum* Labill.					1	
PROTEACEAE	*Adenanthos barbigerus* Lindl.			1			
	Banksia grandis Willd.			1			
	Banksia littoralis R.Br.			1			
	Dryandra bipinnatifida R.Br.					1	
	Dryandra lindleyana Meisn.					1	
	Dryandra sessilis (Knight) Domin	1					
	Grevillea pilulifera (Lindl.) Druce			1			
	Grevillea synapheae R.Br.			1			
	Grevillea wilsonii A.Cunn.			1			
	Hakea amplexicaulis R.Br.			1			
	Hakea cyclocarpa Lindl.			1			
	Hakea lissocarpha R.Br.			1			
	Hakea ruscifolia Labill.			1			
	Hakea undulate R.Br.	1					
	Isopogon sphaerocephalus Lindl.			1			
	Isopogon teretifolius R.Br.			1			
	Persoonia elliptica R.Br.			1			
	Persoonia longifolia R.Br.			1			
	Petrophile seminuda Lindl.			1			
	Petrophile striata R.Br.			1			
	Petrophile sp.1			1			
	Synaphea sp. 1			1			
	Synaphea spinulosa (Burm.f.) Merr.			1			
	Synaphea petiolaris R.Br.			1			
RANUNCULACEAE	*Clematis pubescens* Endl.					1	
	Desmocladus flexuosus (R.Br.) B.G.Briggs & L.A.S.Johnson					1	
RESTIONACEAE	*Desmocladus fasciculatus* (R.Br.) B.G.Briggs & L.A.S.Johnson						1
	Desmocladus sp. 1					1	
	Desmocladus sp. 2					1	
	Desmoscladus sp. 3					1	
	Lepidobolus chaetocephalus Benth.					1	
	Meeboldina coangustata (Nees) B.G.Briggs & L.A.S.Johnson					1	
	Restionaceae sp. 1					1	
	Restionaceae sp. 2					1	
	Restionaceae sp. 3					1	
	Restionaceae sp. 4					1	
	Restionaceae sp. 5					1	
RHAMNACEAE	*Trymalium floribundum* Steud.	1					
	Trymalium ledifolium Fenzl	1					
RUBIACEAE	*Opercularia echinocephala* Benth.			1			
RUTACEAE	*Eriostemon spicatus* A.Rich.			1			
SANTALACEAE	*Leptomeria cunninghamii* Miq.	1					
STACKHOUSIACEAE	*Tripterococcus brunonis* Endl.	1					
STERCULIACEAE	*Thomasia glutinosa* Lindl.						1
STYLIDIACEAE	*Stylidium amoenum* R.Br.		1				

FAMILY	Species	OS Na	OS He	RE Na	RE He	RE Cry	No Gu
	Stylidium brunonianum Benth.		1				
	Stylidium canaliculatum Lindl.		1				
	Stylidium ciliatum Lindl.		1				
	Stylidium diuroides Lindl.		1				
	Stylidium junceum R.Br.		1				
	Stylidium pycnostachyum Lindl.		1				
	Stylidium schoenoides DC.		1				
THYMELEACEAE	*Pimelea spectabilis* Lindl.	1					
	Pimelea suaveolens Meisn.	1					
	Pimelea brevistyla Rye						1
TREMANDRACEAE	*Tetratheca hirsuta* Lindl.	1					
VIOLACEAE	*Hybanthus floribundus* (Lindl.) F.Muell.	1					
XANTHORRHOEACEAE	*Xanthorrhoea gracilis* Endl.			1			
	Xanthorrhoea preissii Endl.			1			
ZAMIACEAE	*Macrozamia riedlei* (Gaudich.) C.A.Gardner			1			

Appendix 4 Spider species found at Mt Dale. Spiders are categorised into guilds; 'H' – Hunting Species, 'W' – Web-Building Species, 'No Gu' refers to the taxa were not classified into either guild.

FAMILY	Species	H	W	No Gu
AMAUROBIIDAE	Amaurobiidae sp. 1		1	
	Amaurobiidae sp. 2		1	
ARANEIDAE	Araneidae sp. 1		1	
	Araneidae sp. 2		1	
CORINIDAE	Corinidae sp.1			1
	Supunna sp. 1			1
CYATHOLIPIDAE	*Matilda* sp. 1		1	
DESIDAE	Desidae sp. 1		1	
DIPLURIDAE	Cethagus ***sp.1***			***1***
GALLENELLIDAE	Gallienellidae sp. 1			1
GNAPHOSIDAE	Gnaphosidae sp. 1	1		
LAMPONIDAE				
	Lamponina loftia *Platnick, 2000*			***1***
	Lamponina sp. 1			1
	Lamponusa gleneagle Platnick, 2000			1
	Longepi woodman Platnick, 2000			1
LINYPHIIDAE	Linyphiidae- sp. 1		1	
	Linyphiidae- sp. 2		1	
LIOCRANIDAE	Licranidae sp.1	1		
MICROPHOLCOMMATIDAE	Textricella sp. 1		1	
	Micropholcommatinae sp. 1		1	
	Micropholcommatinae sp. 2		1	
MITURGIDAE	Miturgidae sp. 1	1		
	Miturgidae sp. 2	1		
	Miturgidae sp. 3	1		
	Miturgidae sp. 4	1		
	Miturgidae sp. 5	1		
	Miturgidae sp. 6	1		
	Miturgidae sp. 7	1		
	Miturgidae sp. 8	1		
	Miturgidae sp. 9	1		
NEMESIIDAE	Anaime ***sp.1***			***1***
OONOPIDAE	*Gamasomorpha* sp. 1	1		
	Myrmopopaea sp. 1	1		
	Opopaea sp. 1	1		
	Orchestina sp. 1	1		
SALTICIDAE	*Lycidas michaelseni* (Simon, 1909)	1		
	Lycidas sp. 1	1		
	Lycidas sp. 2	1		
	Lycidas chrysomelas (Simon, 1909)	1		
	Maratus mungaich *Waldock, 1995*	***1***		

	Hypoblemum sp. 1	1		
	Salticidae Genus 1 sp. 1	1		
	Salticidae Genus 1 sp. 2	1		
	Salticidae Genus 2 sp. 1	1		
	Salticidae Genus 3 sp. 1	1		
SPARASSIDAE	Sparassidae sp. 1	1		
STIPHIDIIDAE	Stiphidiidae sp. 1		1	
THERIDIIDAE	*Dipoena* sp.1		1	
	Dipoena sp. 2		1	
	Gmogola sp.1		1	
	Hadrotarsus sp. 1		1	
	Hadrotarsus sp. 2		1	
	Hadrotarsus sp. 3		1	
	Phoroncidia sp.1		1	
FAMILY	**Species**	**H**	**W**	**No Gu**
	Theridiidae sp. 1		1	
	Theridiidae sp. 2		1	
	Theridiidae sp. 3		1	
THERAPHOSIDAE	Unknown sp. 1			1
THOMISIDAE	Thomisidae- sp. 2			1

Appendix 5 The bird species recorded at Mt Dale. Birds are categorised into guilds; 'T'- Tree foraging, 'G/S'- Ground/Shrub Foraging, 'A' – Air Foraging, 'N' – Nectivore, and 'C' Carnivore. Order as of BIRDSAustralia (2003) which follows Cristidis & Boles (1994).

FAMILY	Common Name	Scientific Name	T	G/S	A	N	C
ACCIPITRIDAE	Brown Goshawk	*Accipiter fasciatus* (Vigors and Horsfield, 1827)					1
	Whistling Kite	*Haliastur sphenurus* (Vieillot, 1818)					1
TURNICIDAE	Painted Button-quail	*Turnix varia* (Latham, 1802)		1			
COLUMBIDAE	Common Bronzewing	*Phaps chalcoptera* (Latham, 1790)		1			
CACATUIDAE	Red-tailed Black-Cockatoo	*Calyptorhynchus banksii* (Latham, 1790)	1				
	Carnaby's Black-Cockatoo	*Calyptorhynchus latirostris* Carnaby, 1948	1				
PSITTACIDAE	Australian Ringneck	*Platycercus zonarius* (Shaw, 1805)	1				
	Red-capped Parrot	*Platycercus spurius* (Kuhl, 1820)	1				
CUCULIDAE	Fan-tailed Cuckoo	*Cacomantis flabelliformis* (Latham, 1802)	1				
	Shining Bronze-Cuckoo	*Chrysococcyx lucidus* (Gmelin, 1788)	1				
STRIGIDAE	Southern Boobook	*Ninox novaeseelandiae* (Gmelin, 1788)					1
HALCYONIDAE	Laughing Kookaburra	*Petroica multicolour* (Gmelin, 1789)		1			
MALURIDAE	Splendid Fairy-wren	*Malurus splendens* (Quoy and Gaimard, 1830)		1			
PARDALOTIDAE	Spotted Pardalote	*Pardalotus punctatus* (Shaw, 1792)	1				
	Striated Pardalote	*Pardalotus striatus* (Gmelin, 1789)	1				
	Weebill	*Smicrornis brevirostris* (Gould, 1838)	1				
	Inland Thornbill	*Acanthiza apicalis* Gould, 1847		1			
	Western Thornbill	*Acanthiza inornata* Gould, 1841		1			
	White-browed Scrubwren	*Sericornis frontalis* (Vigors and Horsfield, 1827)		1			
	Western Gerygone	*Gerygone fusca* (Gould, 1838)			1		
MELIPHAGIDAE	White-naped Honeyeater	*Melithreptus chloropsis* Gould, 1848	1				
	Brown Honeyeater	*Lichmera indistincta* (Vigors and Horsfield, 1827)				1	
	Western Spinebill	*Acanthorhynchus superciliosus* Gould, 1837				1	
	New Holland	*Phylidonyris*				1	

FAMILY	Common Name	Scientific Name	T	G/S	A	N	C
	Honeyeater	*novaehollandiae* (Latham, 1790)					
	Western Little Wattlebird	*Anthochaera lunulate* Gould, 1838				1	
	Red Wattlebird	*Anthochaera carunculata* (White, 1790)				1	
PETROICIDAE	Western Yellow Robin	*Eopsaltria griseogularis* Gould, 1838		1			
	Scarlet Robin	*Petroica multicolour* (Gmelin, 1789)					
NEOSITTIDAE	Varied Sitella	*Daphoenositta chrysoptera* (Latham, 1802)	1				
PACHYCEPHALIDAE	Golden Whistler	*Pachycephala pectoralis* (Latham, 1802)	1				
	Grey Shrike-thrush	*Colluricincla harmonica* (Latham, 1802)		1			
DICRURIDAE	Grey Fantail	*Rhipidura fuliginosa* Sparrman, 1787			1		
CAMPEPHAGIDAE	Black-faced Cuckoo-Shrike	*Coracina novaehollandiae* (Gmelin, 1789)	1				
ARTAMIDAE	Dusky Woodswallow	*Artamus cyanopterus* (Latham, 1802)			1		
	Australian Magpie	*Cracticus tibicen* (Latham, 1802)		1			
	Grey Currawong	*Strepera versicolor* (Latham, 1802)		1			
CORVIDAE	Australian Raven	*Corvus coronoides* Vigors and Horsfield, 1827		1			
HIRUNDINIDAE	Tree Martin	*Hirundo nigricans* Vieillot, 1817			1		
ZOSTEROPIDAE	Grey-breasted White-eye	*Zosterops lateralis* (Latham, 1802)	1				